THE SUCCESS OF THREE RESTORATION PLANTINGS AT KENNEDY'S BUSH, PORT HILLS, CANTERBURY, NEW ZEALAND.

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF PHOTOGRAPHS

ABSTRACT

This study presents the results of an investigation into the success of three different aged (10, 30 and 35 years) restoration plantings at Kennedy's Bush, Port Hills, Canterbury, New Zealand.

Vegetation and ground invertebrates from the three restoration study plots were compared with mature and naturally regenerating forest remnants, and a tussock-grassland study plot using ordination techniques and a variety of diversity indices. Both vegetation and invertebrate communities displayed a developmental sequence from the grassland to the mature forest study plots, suggesting that as the restorations aged they became more similar to the mature forest study plot.

Restoration success is described as a continuum from the recolonisation and establishment of species to the restoration of all ecosystem attributes, including structure, composition and function. The later stages of the continuum cannot occur in the absence of success in the initial stages. Initial species composition at planting and the presence of fruit for attracting birds, features often regarded as essential for early restoration success, did not appear to be critical in this study.

All three restoration plantings at Kennedy's Bush successfully facilitated the recolonisation of native forest plants and ground invertebrates. While the older restoration plantings have restored ecosystem function, all plantings have failed to restore ecosystem structure and composition, suggesting restoration has not successfully restored ecosystem structure and function yet.

However, the future of the plantings looks promising. It is suggested that as the plantings age they will more closely resemble the mature forest community at Ahuriri Scenic Reserve and should successfully restore ecosystem structure as well as function, indicating that future restoration projects in the study area are likely to be successful.

CHAPTER ONE

INTRODUCTION, BACKGROUND and STUDY AIMS

Ecological restoration provides an opportunity to return an ecosystem to a close approximation of its condition prior to disturbance (National Research Council 1991). Atkinson (1988) defines restoration as "active intervention and management to restore or partially restore biotic communities, both their plants and animals, and the associated physical environment as fully functioning and sustainable systems". It is these two definitions, with the implication that restoration provides an opportunity to compensate for past environmental degradation which provide the basis for this study.

There appears little need to emphasise the extent of habitat destruction and modification, and the implication of this for human civilisation. Concern for the diminishing amount of natural area and resources has led to a tremendous increase in interest in restoration as a technique for reversing world-wide habitat degradation (Bradshaw 1983; Jordan *et al* 1987; Cairns 1988; Jordan *et al* 1988; Saunders *et al* 1993; Bowles & Whelan 1994). While there are still many practical problems associated with restoration ecology, it has been hailed by some as a new paradigm for biological conservation (Hobbs & Norton 1996).

Like many branches of ecology, ecological restoration appears to have been branded with a variety of terms including restoration, rehabilitation, reallocation to reconstruction (Hobbs & Norton 1996). While there are subtle differences in meanings, which has been the source of considerable debate in the restoration literature, they are often used interchangeably in many publications. Hobbs & Norton (1996) suggest that the endless quibbling over terminology distracts from "the work at hand", wasting valuable time. I feel that while the diversity of restoration projects may have a diversity of goals associated with the nature of the restoration (hence the diversity of terms), the basic approach is essentially the same for all projects. The term restoration is used here to encompass all forms and levels of restoration.

In addition to restoring degraded ecosystems, restoration has considerable potential in other aspects of conservation biology. Restoration is of direct use in conservation through the provision of additional habitat, creating of buffer zones and linking of fragments by establishing corridors (Hobbs 1993). Berger (1990) even suggests restoration may have the potential for controlling non-native species by altering the physical properties of habitats and enhancing the availability of biological control agents to produce ecological conditions that are inhospitable to invaders. In many parts of the world the extent of loss of some habitats means that conservation of these systems is no longer possible. For example, New Zealand has many habitat types that are not well represented in protected areas (O'Connor *et al* 1990). Restoration may be the only means available to enhance representativeness (Awimbo *et al* 1996; Hobbs & Norton 1996). While restoration provides an opportunity to restore highly degraded sites, restoration is also valuable in mitigating the effects such degradation has on soils and the environment (Saunders *et al* 1993). Hobbs & Norton (1996) suggest restoration activities may occur along a continuum from the complete rebuilding of totally devastated sites (such as mine sites) to limited management of relatively unmodified sites (eg. removal of pest species). Restoration projects do not have to be constrained by scale and limited to particular areas (such as mined sites). There is increasing recognition of the value for restoration at greater scales of organisation, particularly at the landscape scale (Hobbs 1994; Naveh 1994). Restoration also provides an excellent opportunity for local people to become involved in local land management issues (Cairns 1993b; Saunders *et al* 1993).

While restoration is clearly a valuable technique there has been much criticism, especially ethical, of restoration practises (eg. see Elliot 1982,1994; Gunn 1991; Cowell 1993; Ehrlich 1993; Fry & Main 1993; Jordan 1993; Scherer 1994). In particular, if restoration is capable of returning a degraded site to its prior condition then restoration may be (and is in some cases) used to justify habitat modification (Elliot 1982; Gunn 1991; Higgs 1993). Much of the argument surrounding this ethical problem involves whether or not restoration can restore naturalness. Gunn (1991), Cowell (1993) and Elliot (1982,1994) illustrate the philosophical complexities surrounding this issue, being immersed in discussion circling the inherent value and authenticity of restored systems, relative to natural systems. Perhaps Jordan (1993) best illustrates the practical value of this ethical debate to the actual practice of restoring degraded lands by suggesting that while we may make a forest look as good as the original it won't "sound as good, and won't smell as good, and it won't have any ghosts in it".

Restoration ecology has also been criticised as not being a true science (Cairns 1991; Bradshaw 1993, 1994). Restoration until recently has largely progressed on an ad hoc, site- and situation- specific basis with little development of general theories or principles (Hobbs & Norton 1996). This is illustrated internationally by Majer & Recher (1994) who showed little exchange of ideas in the restoration literature between different geographical areas. It is important for the development of restoration ecology as a science that each restoration project is strictly monitored and published so future projects may benefit from the experiences of the past (Bradshaw 1993; Majer & Recher 1994; Hobbs & Norton 1996). Lessons learnt from failed projects are as important as those learnt from successes (Bradshaw 1993). Projects that fail due to inadequate preparation are a waste of time and resources and may discourage enthusiasm for future restoration projects (Hobbs & Norton 1996).

There is currently renewed interest in how ecological systems may assemble (Drake 1990; Drake *et al* 1993). Such assemblage rules are of value to ecological restoration. Furthermore, restoration ecology may provide the opportunity to test these and other ecological theories. As Bradshaw (1987) suggests "The acid test of our understanding is not whether we can take ecosystems to bits on paper, however scientifically, but whether we can put them back together in practise and make them work". Restoration may provide the ultimate opportunity to examine our current understudying of ecological principles.

One aspect of restoration that has been identified as centrally important is monitoring the development and success of restoration projects (Berger 1991; Cairns 1991; Norton 1991; Westman 1991; Aronson *et al* 1993a; Morrison 1994). The need to monitor projects has a very practical base. Mining companies are often required to pay large financial bonds, to be released when the site is returned to its former condition (Norton

1991). At what point is it possible to say that a project has been successful and that the bond should be released?

In the absence of predetermined goals it is not possible to assert whether a restoration project is successful or not. Restoration goals may be perceived on a continuum from the return of social and aesthetic values at one end, to the complete restoration of a fully functional self-sustaining natural system at the other. In the past, goals have focussed on the restoration of a natural system. However there has been some confusion regarding what's considered natural. Some ecosystems have been modified for long periods. For example, in New Zealand is the natural state pre-human or pre-European? (Hobbs & Norton 1996). Using a term such as "natural" implies that vegetation would remain in a static state in the absence of intervention (Pickett *et al* 1992; Pickett & Parker 1994). The way in which we view ecosystems has changed from where we see them as being static and predictable to where they are now viewed as being dynamic and changing (Pickett & Parker 1994). Furthermore, due to extinctions and invasions it is in most cases impossible to restore ecosystems to past conditions, hence Hobbs & Norton (1996) consider it naive to attempt to restore to specified historical conditions.

Cairns (1989, 1991) and Aronson *et al* (1993a) suggest efforts to restore ecosystems to a defined indigenous system, resembling the original, handicaps projects by placing goals on them that they are unlikely to achieve. Hobbs & Norton (1996) suggest that reference systems are required to guide restoration planning provided they are based on similar landform/soil/biotic/climatic conditions. However, they do not believe that reference systems should be used as goals for restoration efforts. They suggest that this is unrealistic, unachievable and static (Hobbs & Norton 1996). Goals that are dynamic and consider the changing nature of the environment are thought to be more realistic.

Cairns (1989,1991), Aronson *et al* (1993a) and Aronson *et al* (1995) support the idea that some sort of reference system is required for restoration projects. Aronson *et al* (1995) while agreeing with Pickett and Parker's (1994) dynamic views on ecosystems, suggest that in the 'real-world' restoration projects require an ecosystem of reference. While it is often not clear what should be used for the ecological 'yardstick', it is desirable to establish some standard of comparison and evaluation, even if arbitrary and imperfect (Aronson *et al* 1995). In the absence of reference or control Aronson *et al* (1995) suggest an experiment cannot be evaluated. While recognising that there may be a large number of ecosystem trajectories worth considering, enough flexibility to modify goals in light of the first few years of results will ensure that the original ecosystem of reference, or "some standard of comparison and evaluation, even if the choice is made is somewhat arbitrary" (Aronson *et al* 1993a) will remain a useful tool.

In light of Cairns (1989, 1991), Aronson *et al* (1993a) and Aronson *et al* (1995), I suggest that the use of an ecosystem of reference is valuable to monitor and indicate those species and processes that may be expected to be present in a restored system. However, I stress that I do not consider such a restoration project to be a failure if for example, exact species compositions are not found in comparison with the reference system. While I feel that the use of an ecosystem of reference is appropriate for this study, I am aware of potential associated problems and use such a reference system cautiously. It is unrealistic to expect restoration data to be 100% similar to reference systems (Cairns 1991; Westman 1991). Due to the difficulties associated with evaluating the success of systems (Berger 1991; Westman 1991; Aronson *et al* 1993a) I suggest that the careful use of an ecosystem of reference, with a "lid" on perceived expectations is required to provide indications of what we may look for to suggest whether a restored system has reached predetermined goals.

Each restoration project provides the opportunity to test methods or ideas that may be valuable to future studies. Most studies monitoring restoration progress do so after 5 or so years (Chambers *et al* 1994). The present study provides a unique opportunity to evaluate progress after significantly longer periods of restoration development. In the process of evaluating the success of the plantings in this study I endeavour to draw attention to some ideas that have been proposed and make attempts to evaluate them for the benefit of further restoration work.

The objectives of this study are;

(1) To evaluate whether the three restoration plantings are facilitating the colonisation and establishment of native forest plant and invertebrate species.

(2) To determine whether those plant species used in the initial plantings influence the subsequent colonisation and establishment of native forest species.

(3) To determine if the planting of fruiting tree species is required to attract birds.

(4) To determine the role, if any, of non-native species in restoration.

(5) To determine whether the restoration plantings in this study have been successful.

CHAPTER TWO

STUDY AREA

This chapter provides a brief summary of the geology, landforms, climate and vegetation of the study area. An overview of the historical and post-human modifications of the study area and a discussion of restoration goals is also given.

2.1 Location.

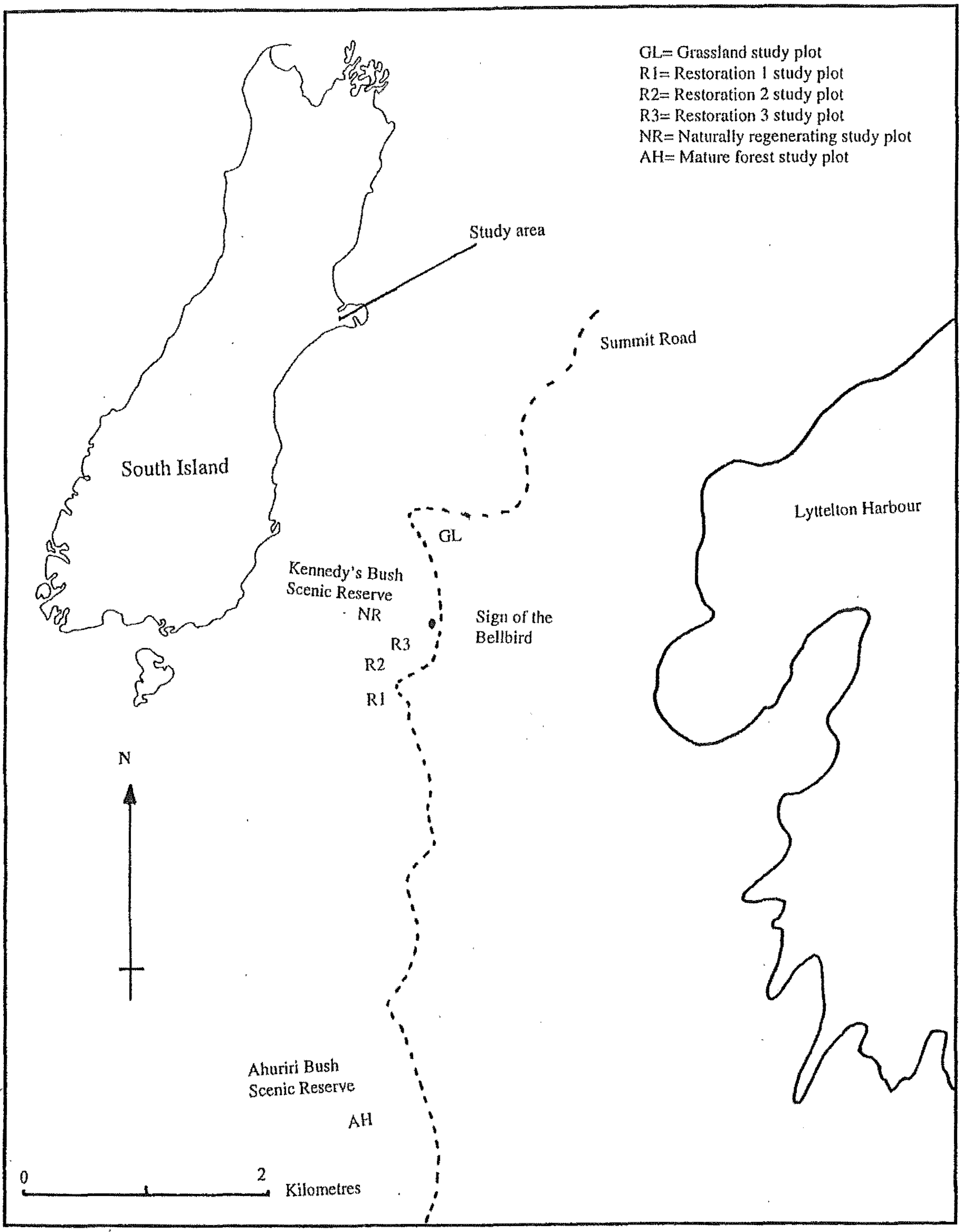
This study was carried out on the Port Hills, overlooking the city of Christchurch to the north west and Lyttleton Harbour and Banks Peninsula to the south east. The study area is within the Port Hills Ecological District, a part of the Banks Ecological Region (Wilson 1992). Three sites at the south west end of the Port Hills were studied. The first site was a grassland covered knoll, the second site was at Kennedy's Bush Scenic Reserve, an area of restoration planting and native regeneration and the final site was at Ahuriri Scenic Reserve, an area of remnant mature forest. The three study sites are within 5 km of each other, and are all in close proximity to the Summit Road, which runs across the top of the Port Hills from Sumner to Gebbies Pass (Figure 2.1, Table 3.1).

2.2 Geology and Landform.

The Port Hills, reaching a maximum altitude of 573 m at Coopers Knob, rise steeply from Lyttleton Harbour on the southern and eastern margins and more gently from the Canterbury plains on the northern and western margins. Providing contrast to the vast expanse of the Canterbury Plains, the Port Hills abut the Pacific Ocean, the Avon Heathcote Estuary and Christchurch City (Wilson 1992).

Formed during the activity of the Lyttleton Volcano between 11 and 10 million years ago (Sewell 1985), the Port Hills represent the north western to south western side of the eroded caldera centred on Lyttleton Harbour (Griffiths 1974). The rocks are of basaltic

Figure 2.1 Map showing the location of the study plots.



and trachyte material from the Lyttleton eruption, as well as younger, local outcrops of olivine-rick basalts formed during the Diamond Harbour Volcanics, 6 million years ago (Weaver *et al* 1985). The Lyttleton Volcano is characterised by low angle cones (shield volcano) and was a 'Hawaiian' style eruption, where most activity was relatively quiet. The lava flows are mostly of the aa type, and the radial dyke swarm is one of the best developed and exposed in the world. After the eruptions ceased, the summit, estimated to have been 1500 m in height, eroded quickly to its present relief (Weaver *et al* 1985).

The Port Hills are partly covered by a mantle of loess, the thickest deposits being found on low elevation north facing slopes (up to 270 m) although thick deposits can occur on rolling tops at higher elevations. However, steep slopes at high elevations generally have little or no loess. Loess was mainly deposited during late Pleistocene glacial periods and is likely to have completely covered the Port Hills at times. The peri-glacial conditions that existed at this time (frost lift and freeze thaw) would have initiated erosion almost as soon as the loess was deposited (Griffiths 1974), leading to the present uneven distribution pattern.

2.3 Soils.

The soils of the Port Hills are derived from igneous rock and loess. Those soils on the lower slopes are mainly yellow-grey earths, while the more upper slopes are characteristically yellow-brown earths, brown granular loams, or a mixture of the two. Small areas of recent alluvium are present at the mouths of some valleys and include gley recent soils, saline gley recent soils, organic soils and yellow-brown sands (Wilson 1992).

The soils of the three study sites belong to the Rapaki-Summit hill soil complex. This intergrade soil has a colluvium of basalt (Rapaki soil) and loess (Summit) and is made up of 25% each of Rapaki and Summit hill soils and 50% of intergrades between the two. This soil complex is classified as an intergrade between brown granular loams and yellow brown earths, or technically, a moderately to strongly enleached elfulvoprospadic. Both overall and internally this soil type is well drained (Griffiths 1974) and becomes dry during late summer months, especially on north facing slopes (Williams 1983).

These soils are of moderate fertility and acidity (pH around 5.5). While low in amounts of available phosphorous there are moderate amounts of exchangeable cations. Available nutrients increase with intermixing of the basalt stones (Fitzgerald 1966).

2.4 <u>Climate.</u>

The average annual temperature in Christchurch City is 11.6 °C . July is the coldest month, with a mean daily temperature of 10.2 °C. During summer (January-February) 20% of the days are over 25 °C and temperatures over 30 °C are not uncommon. The greatest variability in temperature occurs during the winter months (July). The close proximity of the study area to the ocean means a moderating effect on temperature extremes, as well as contributing to both sea breeze and sea fog effects. Christchurch averages 1985 hours of bright sunshine per year (McGann 1983). Temperature normals may be estimated for areas where climatic monitoring is not available using multiple regression equations (Norton 1985). Using this method the mean summer and winter temperatures at the three study sites were calculated as 13.5 ± 0.7°C and 5.2 ± 0.5°C respectively, suggesting the study area is significantly cooler than Christchurch City reflecting their higher altitude.

Wind patterns across the South Island are greatly influenced by the Southern Alps. The prevailing winds at the study area come from the east-north east and south west quarter. These are the dominant rain bearing winds. 50% of the annual rainfall is associated with south west winds (Jayet 1986). North west winds are infrequent, yet significant, especially in the higher wind speed range. These winds are characteristically associated with high temperatures and relatively low humilities in summer months, and give the highest temperatures in all seasons. Sheltering from Banks Peninsula means that south east winds are not frequent. 50 % of winds in Christchurch are of a speed between 5 and

19 km/hr. 20 % are in a wind speed range of 20-30 km/hr. Less than 10 % of winds are strong, while calm conditions are experienced 17 % of the time (McGann 1983). Being elevated the crest of the Port Hills is considerably windier than Christchurch.

The upper reaches of the Port Hills from Kennedy's Bush to Ahururi Scenic Reserve have a mean annual rainfall of 800-900 mm, the wettest month being July, and the driest February. The Hills at the western end of Lyttleton Harbour tend to experience higher rainfalls, the result of being exposed to two moisture bearing winds (easterlies and southerlies) being forced to flow over the Hills at a similar location (Jayet 1986).

The Port Hills experience some snow and hail during winter months, and frosts, associated with calm periods, are common in winter.. Periods of drought, or little rain during summer months are also common (Jayet 1986) and can be a significant ecological factor (Innes & Kelly 1992).

The study area lies in the lowland (up to 500m) to lower montane (500-700 m) (Wilson 1992) bioclimatic zone. Jayet (1986) describes this region as being humid.

2.5 Vegetation Pattern.

Prior to Polynesian arrival the Canterbury landscape was almost completely forested (Molloy *et al.* 1963, Molloy 1969). Evidence suggests that there was complete forest cover below the climatic tree line over virtually the whole of the South Island except Central Otago. The distribution of the dominant forest species in these times is unlikely to have differed from the distributions of their present remnants (Molloy 1969). At this time the Port Hills would have been predominantly forested with podocarp-mixed-angiosperm forest being dominant. This forest canopy was characterised by lowland totara (*Podocarpus totara*), matai (*Prumnopitys taxifolia*), kahikatea (*Dacrycarpus dacrydiodes*), mahoe (*Melicytus ramiflorus*), broadleaf (*Griselinia littorals*), pepperwood (*Pseudowintera colorata*), kaikomako (*Pennantia corymbosa*) and pidgeonwood (*Hedycarya arborea*). Montane forest, characterised by thin bark totara (*Podocarpus *hallii*), broadleaf, pepperwood and softleaved tree fern (*Cythea smithii*), would have appeared on higher ground. Bluffs would have been vegetated with non-forest species such as snow-tussock (*Chionchloa rigida.*), silver tussock (*Poa cita*) and *Dracophyllum* species (Wilson 1992).

The first humans came to New Zealand around 950 AD. These eastern Polynesian people did not practise agriculture. They were however hunters (especially of moa) and used fire in their economy. Largely through accidental burning these people were responsible for the first extensive transformation of the South Island landscape over a 500 year period (Johnson 1969). The first introduced mammals, kuri (Polynesian dog) and kiore (Polynesian rat) arrived with these people. By the end of the first 500 years of settlement one third of the forest cover of the Banks Peninsula was removed by fire and 30 species of bird, including moa and other large ground dwelling birds, had reached extinction (Wilson 1992). The Ngai Tahu were the first Banks Peninsula people living according to a "classic" culture, thought of today as being Maori. These people arrived from the North Island in the 17th century (Wilson 1992). Although these early inhabitants had a very significant effect on the extinction of much fauna and cleared a large proportion of forest, the number of plant species lost was low, or nil (Wilson 1992).

Within the last 1000 years much of the forest on the northern slope of the Port Hills would have been removed. Large areas appear to have been under native tussock (or a fire maintained tussock-grassland-shrubland mosaic) for centuries, largely as a result of Polynesian fires. Such vegetation would have appeared completely natural to the first European colonists 150 years before present.

The impact of Europeans from the early 1800's was severe. Finding a largely forest dominated landscape they began to exploit it immediately. By the early 1900's virtually all the Banks Peninsula forest was milled or burnt (Johnson 1969), and the remains ravished by fire followed by introduced mammals and plants dramatically changing a once natural environment (Molloy 1969). The present vegetation of the Port Hills is a great modification on what the early European settlers would have observed 150 years ago. Much of the area is now grazed by sheep and other introduced mammals. Exotic forestry and urban settlement are a dominant feature, while only small fragments of old growth forest remain. These and a moderately large area of regenerating mixed angiosperm and kanuka (*Kunzea ericoides*) dominant second growth forest are constantly threatened by introduced pests such as possums, goats and rats, and by fire. While a large amount of tussock grassland remains, this is thinned to a large degree by introduced herbs and grasses and does not resemble the dense silver tussock grassland present in the early 1800's (Wilson 1992).

2.6 History of the restoration plantings at Kennedy's Bush.

As the main focus of this study is based on the success of some of the restoration plantings at Kennedy's Bush, it is important to provide some background on this area. In this section I discuss some of the biological and social implications of European modification of the Kennedy's Bush natural area and provide a brief history as to how and why Kennedy's Bush became a site for restoration.

Kennedy's Bush has a long and complex post-European history. Early demands for timber and firewood by the first settlers in Christchurch meant that supplies on the flat (where Christchurch now stands) were quickly exhausted (Ogilvie 1978). The Port Hills were then utilised for wood to satisfy these demands (McCaskill 1978).

Milling commenced first in Hoon Hay Valley in 1852. In 1856 Thomas Kennedy bought a section of 28 acres at the head of Landsdown Valley, hence the name Kennedy's Bush. Before milling commenced there was 120 hectares of virgin bush. Kennedy set up a business cutting firewood and fence posts and later farmed further down the valley (McCaskill 1978, Ogilvie 1978). Much of the bush was cut by two brothers Forster who made their homes there (Baughan *et al* 1914). The best timber was cut from this area by 1870. This included totara, matai, kahikatea, broadleaf and fuchsia, although a huge fire in 1868 destroyed much of the forest in Kennedy's Bush and Hoon Hay valley. By 1900 almost all of Hoon Hay and Burkes Bush had been destroyed but Kennedy's Bush was

Photo 1 Kennedy's Bush prior to 1928 (from Cowan 1928).

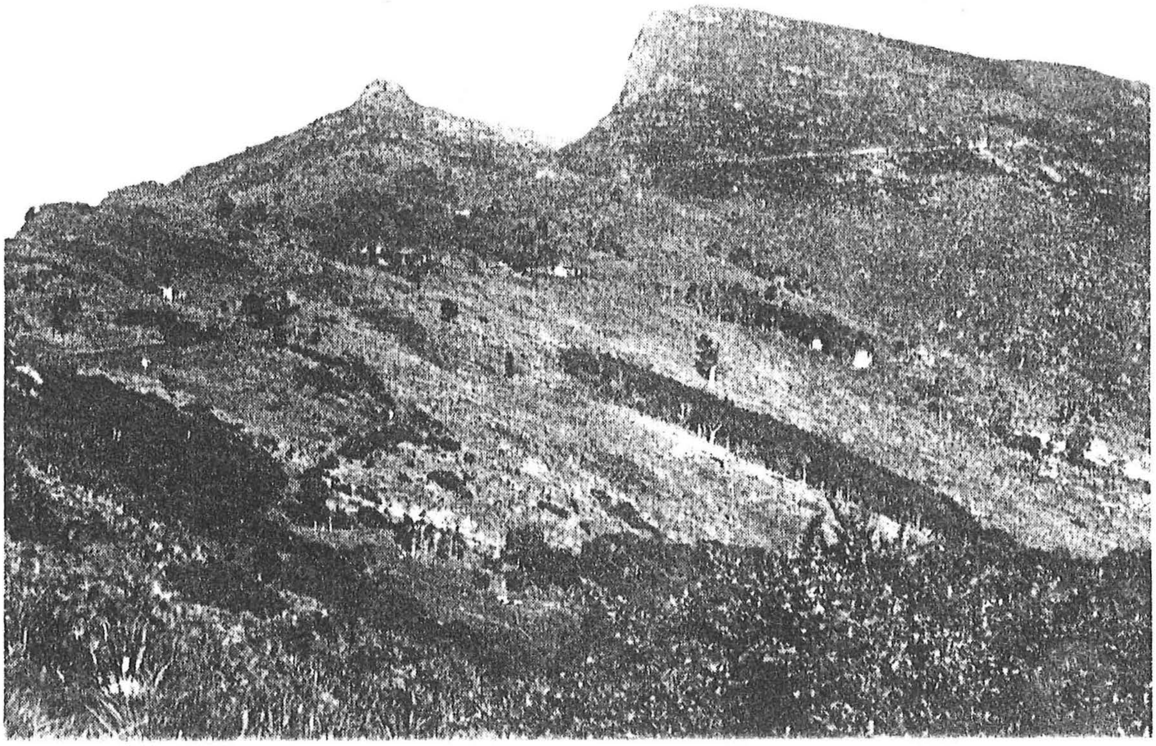
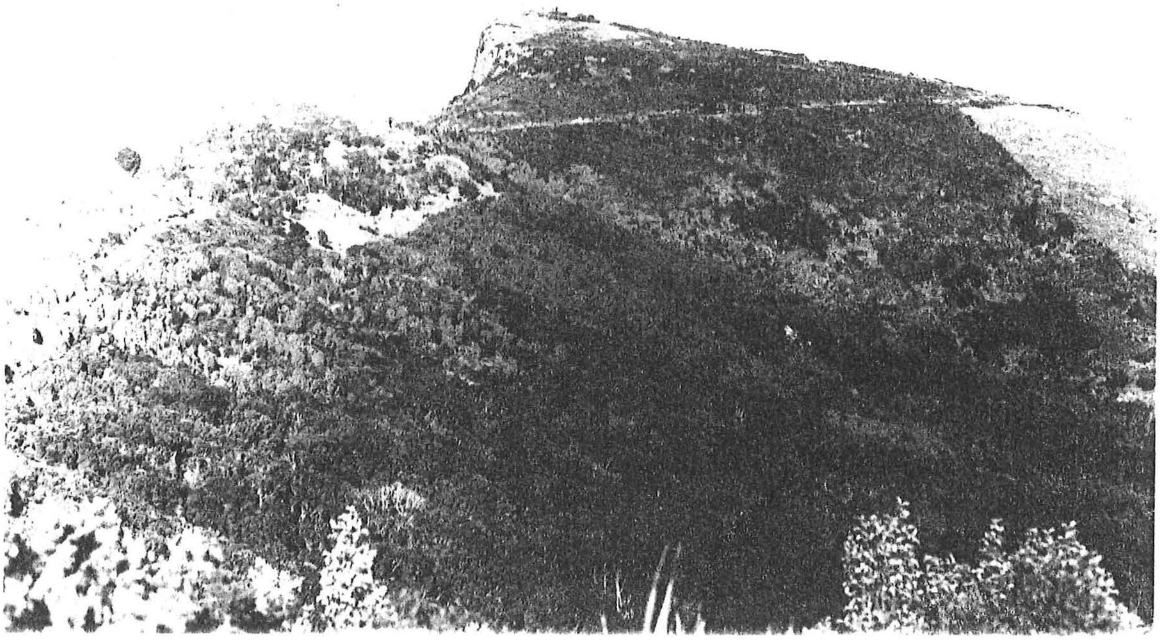


Photo 2 Kennedy's Bush in 1995.



still considered by H.G. Ell to be worth saving (McCaskill 1978).

In 1900 Ell rode to Kennedy's Bush via Dyers Pass with Wm. Reece (Mayor of Christchurch) and Albert Loe (the owner) to discuss the purchase of 20 ha of bush and scrub so it could be secured against further destruction and provide a sanctuary for native birds. This was the beginning Ell's conservation career. 1906 saw Cockayne and Ell listing 136 species of native plants from Kennedy's Bush (McCaskill 1978). By 1908, 20 hectares of Kennedy's Bush had been purchased with the help of a Government subsidy and gazetted under the Scenery Preservation Act (Ell's most notable achievement as a Member of Parliament was the passing of this Act in 1908) (Ell 1929, Olgivie 1978).

In 1908 the Kennedy's Bush Scenic Board was appointed to control the new reserve. As soon as this newly formed board met, Ell and Cockayne persuaded the rest of the committee that the remainder of Kennedy's Bush should be acquired (Ogilvie 1978). Up until this time Ell had been planning the idea of a road along the summit of the Port Hills and the rest of Banks Peninsula, which was to face strong opposition until the late 1920's. The first sod was turned in 1908, by the Governor, Sir Charles Bowen. Soon after Ell formed the Summit Road Association, which had a close association with the Kennedy's Bush Scenic Board. The summit road association was to undergo a few changes before becoming the Port Hills-Akaroa Summit Road Public Trust in 1925. This trust worked toward forming and maintaining the summit road, acquiring areas of native bush to form bird sanctuaries, forming walking tracks and building stone refreshment houses for those who used the summit road (Ell 1934).

Fire damage to the bush in 1911 and 1912, resulted in a resident caretaker taking up residence. A stone cottage was built, using stone quarried on site, for the caretaker Mr D. Potter and his wife, and to provide refreshments for visitors (Andersen 1927; Ogilvie 1978). The Sign of the Bellbird, as it was called was first occupied in 1913. The reserve at this time was securely fenced and a grazing lease for sheep only had been arranged (McCaskill 1978). Ell gave the stone house at the Sign of the Bellbird the Maori name Orongomai. This came from the Ngai-tahu name for Cass Peak, meaning "the place where voices are heard".

In 1915 Cockayne, noted Kennedy's Bush as the most important remaining example of the forest class which originally occupied the gullies and hollows of the Port Hills. No longer in virgin condition the reserve contained examples of probably all flowering plants and most ferns which formed the original forest. Cockayne (1915) suggested that Kennedy's Bush was an important natural museum.

In October 1931, a serious fire swept through the major part of Kennedy's Bush destroying approximately 20 ha. Most damage was to young trees and seedlings and the old part of the bush below the Sign of the Bellbird was not affected (Scotter 1965, McCaskill 1978). Flames were reported to be 6-18 m high (Ell scrapbook).

In the years after Ell's death (1934) and the war (1939) there was considerable dissatisfaction with the control of the Port Hills reserves, particularly the condition of Kennedy's Bush. Fences had fallen down and damage by stock was severe. The board seemed to encourage grazing as a means of generating revenue. Unable to put up with continuous criticism the board refused to carry on and the Lands Department gained control of the reserves (McCaskill 1978), which were later transferred to the Christchurch City Council.

In 1947 McCaskill (1978) visited some of the reserves showing they were in a considerable state of neglect. In 1952 the initiative was taken to repair Kennedy's Bush and in 1953 Kennedy's Bush was fenced. M.J. Barnett, Superintendent of Reserves appealed for voluntary help with weed control in 1953 and planned to follow this with the planting of nursery raised trees and shrubs indigenous to the area. The first 2000 trees and shrubs were provided by the Christchurch City Council Reserves Department and were planted on arbour day August 1953. Since then regular planting and clearing round young tress has gone on (McCaskill 1978), although these are largely undocumented.

2.7 Goals for the restoration at Kennedy's Bush.

To evaluate the success of three restoration plantings at Kennedy's Bush, three reference systems are used: Ahuriri Bush, a remnant patch of native lowland podocarp forest is used as a model to indicate those species and process that occur naturally; A naturally regenerating remnant forest patch at Kennedy's Bush is used to indicate those features characteristic of a later successional natural system; A grassland area near Kennedy's Bush is used to indicate the likely starting point for the restoration plantings in this study. The use of such a system in evaluation provides a firm base from which to monitor potential restoration development.

I am unaware of any specific goals having been set for the restoration plantings at Kennedy's Bush. In the absence of goals it is not possible to determine whether a restoration project is successful, as success is reached when some predetermined goal has been reached. Consequently, I have devised goals for these plantings that are likely to be similar to what people involved in the planting would have suggested. I suggest that the Kennedy's Bush restoration was initiated out of concern for the diminishing amount of native forest ecosystems in the Port Hills area. While I would not be so naive as to imply that these plantings should restore the Kennedy's Bush area to exactly as it was prior to modification, I feel that it is appropriate to expect certain elements of the previous system to be present in a successful restoration planting, as they are still present in largely intact remnant systems in the same area.

An appropriate goal for the restoration plantings in this study might therefore be to restore fully functioning self-sustaining systems that are ecologically appropriate for the area and, in doing so, enhancing opportunities for native biodiversity conservation. While such a goal appears rather abstract it demands those features necessary for functioning natural systems to be present, while not constraining the restorations with a goal which is to restrictive. If it is ecologically appropriate for the restorations to be similar (in appearance, function, etc.) to the natural systems then they will. If the restored systems develop to slightly different states from the selected reference ecosystems, but are self-sustaining with a predominance of native species, then they are no less successful due to the flexibility of the proposed goal. Ultimately this goal should see the restoration of native forest, within the context of the changes that have occurred due to the consequences of human and especially European colonisation

CHAPTER THREE

METHODOLOGY

This chapter describes the methodology used in this study and summarises the main features of each study plot.

3.1 Site selection and location.

The overall aim of this study was to assess whether restoration plantings facilitate the recolonisation of native flora and invertebrates. Six study plots were subjectively chosen to encompass a range of restoration development. Study plot selection was constrained by needing to have similar aspect, slope and altitude, and to have plots located within 5 km of each other (Figure 2.1, Table 3.1). These constraints were necessary to reduce any environmental variation between study plots that may influence the results.

The grassland study plot was chosen as it is likely to be similar in vegetation and site conditions to the restoration sites prior to restoration planting. The restoration plots were subjectively chosen based on their relative age; a young planting, an older more established planting, and an intermediate one. These three study plots are located in close proximity to each other at Kennedy's Bush, and appear to have undergone similar planting treatments. The naturally regenerating study plot, also at Kennedy's Bush, was chosen to compare its flora and faunal development with the restoration plantings. The mature forest study plot at Ahuriri Scenic Reserve was chosen as an example of what the forest may have been like at Kennedy's Bush had it survived. It is thought that most species dependent today on historic Port Hills forest habitat are likely to be found in Ahuriri Scenic Reserve, as it is considered to be the best and only example of unmodified natural forest in this area (Kelly 1972; Wilson 1992). Due to the extensive human modification of the study area it is unlikely that forest species that have failed to find refuge in Ahuriri Scenic Reserve would have escaped extinction from the general area.

Table 3.1 The three study sites and the six study plots they contain.

	Grid reference	Altitude	Aspect	Slope
Grassland				
GL	M36 796413	460 m	NNW	11°
Kennedy's Bush				
NR	M36 795307	400 m	W	25°
R3	M36 795307	420 m	W	25°
R2	M36 795307	440 m	SSW	25°
R1	M36 795307	440 m	NNW	25°
Ahuriri S.R.				
AH	M36 796267	440 m	SSW	20°

GL= Grassland study plot
R1= Restoration 1 study plot
R2= Restoration 2 study plot
R3= Restoration 3 study plot
NR= Naturally regenerating study plot
AH= Mature forest study plot

All study plots were within easy access from the main Summit Road, although accessibility was not intentionally a component in study plot selection. The mature forest study plot at Ahuriri Scenic Reserve is 4.5 km from Kennedy's Bush and the grassland study plot. This study required that I was able to access all study plots within one day in order to carry out the invertebrate sampling. Due to the close proximity of the study sites there was never any difficulty in this.

3.2 Vegetation description.

Grassland (GL).

The dominant species in this study plot is the introduced pasture grass cocksfoot (*Dactylis glomerata*). The native silver tussock (*Poa cita*) and bracken (*Pteridium esclentum*) are also common. Vetsch (*Vicia sativa*), hard tussock (*Festuca novae-zealandiae*), browntop (*Agrostis capillaris*), yorkshire fog (*Holcaspis lanatus*), clover (*Trifolium sp.*), sweet vernal (*Anthoxanthum odoratum*), chewings fescue (*Festuca rubura subsp. commutata*), sheep spur (*Acanena anserinifola*) and other tussock/grassland herbs and grasses are also present.

The structure of this study plot is typical of grassland in the study area, a dense, up to 1 m tall area of silver tussock and bracken.

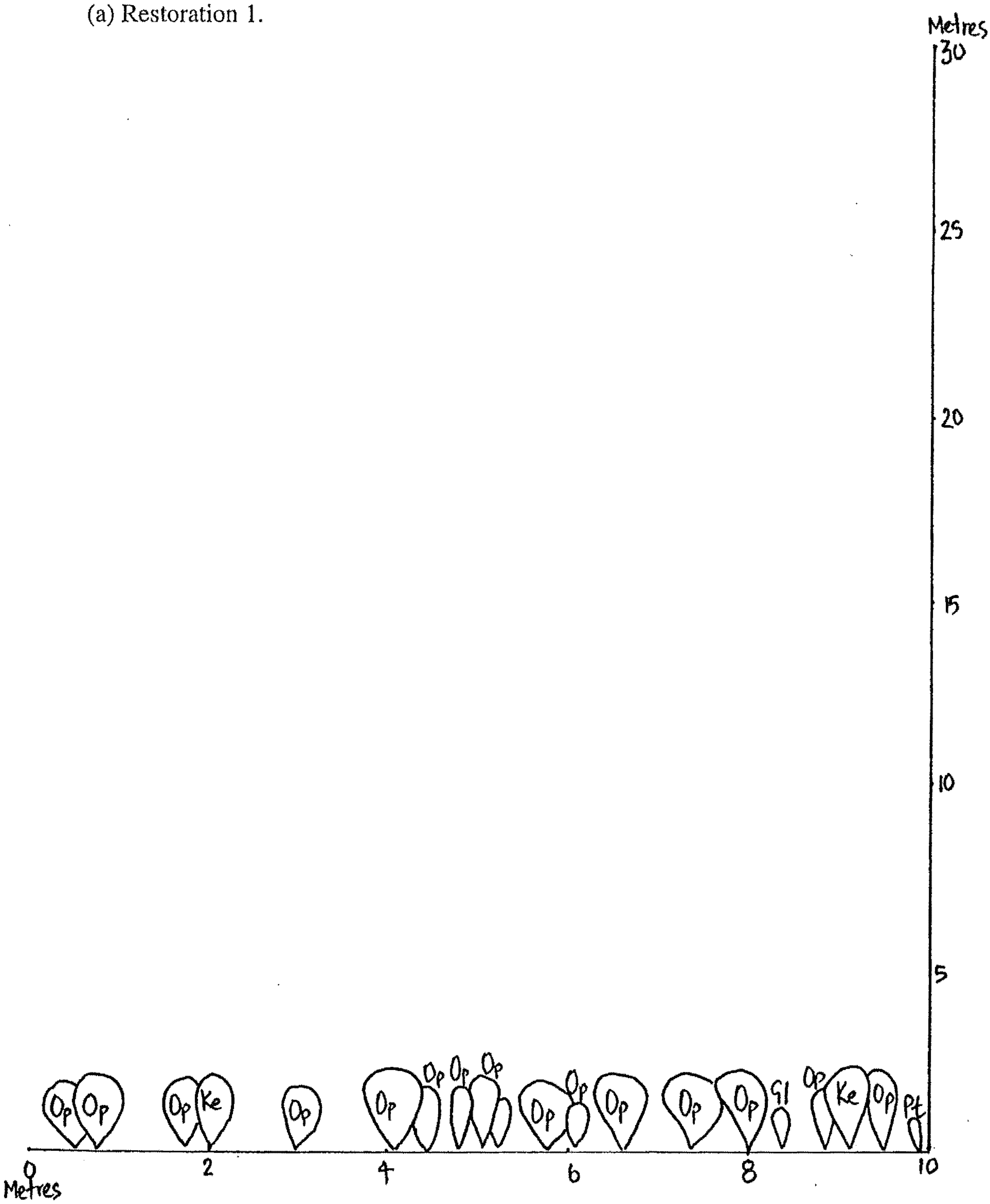
Restoration 1 (R1: Figure 3.1a, Table 3.2).

This planting is approximately 10 years old. The dominant shrub species in this study plot is the planted *Olearia paniculata*. Kanuka (*Kunzea ericoides*), *Plagianthus regius* and kowhai (*Sophora microphylla*) have been planted in low numbers into a tussock/grassland floristically similar to the grassland study plot.

The grassland component of this study plot is structurally similar to the grassland study plot. The tree species have been planted both singly and in clumps of up to 5 m^2 and are

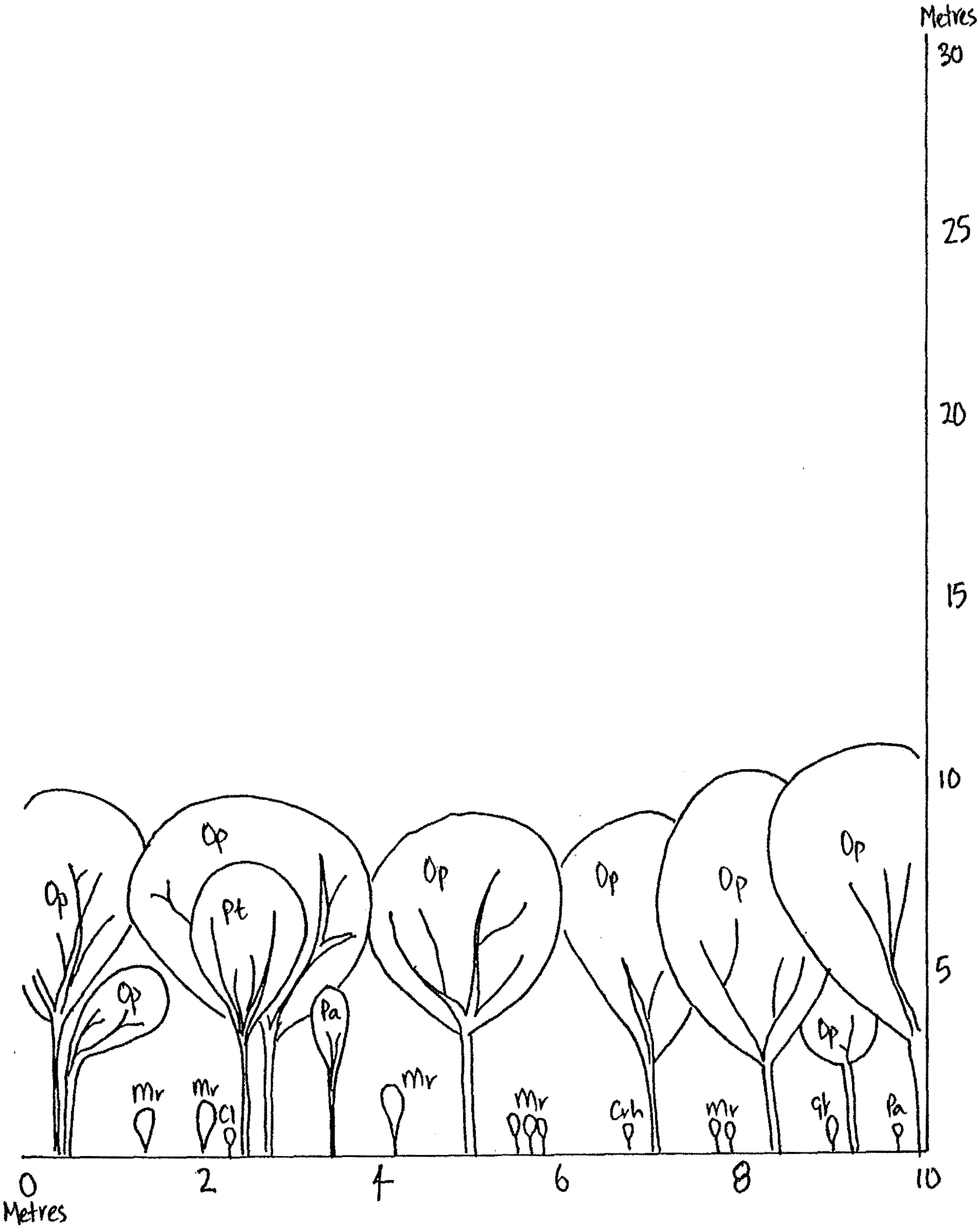
Figure 3.1 Profile diagrams. Figures show vertical profile diagrams of the three restoration, naturally regenerating and mature forest study plots (based on a 10 x 2 metre transect).

(a) Restoration 1.



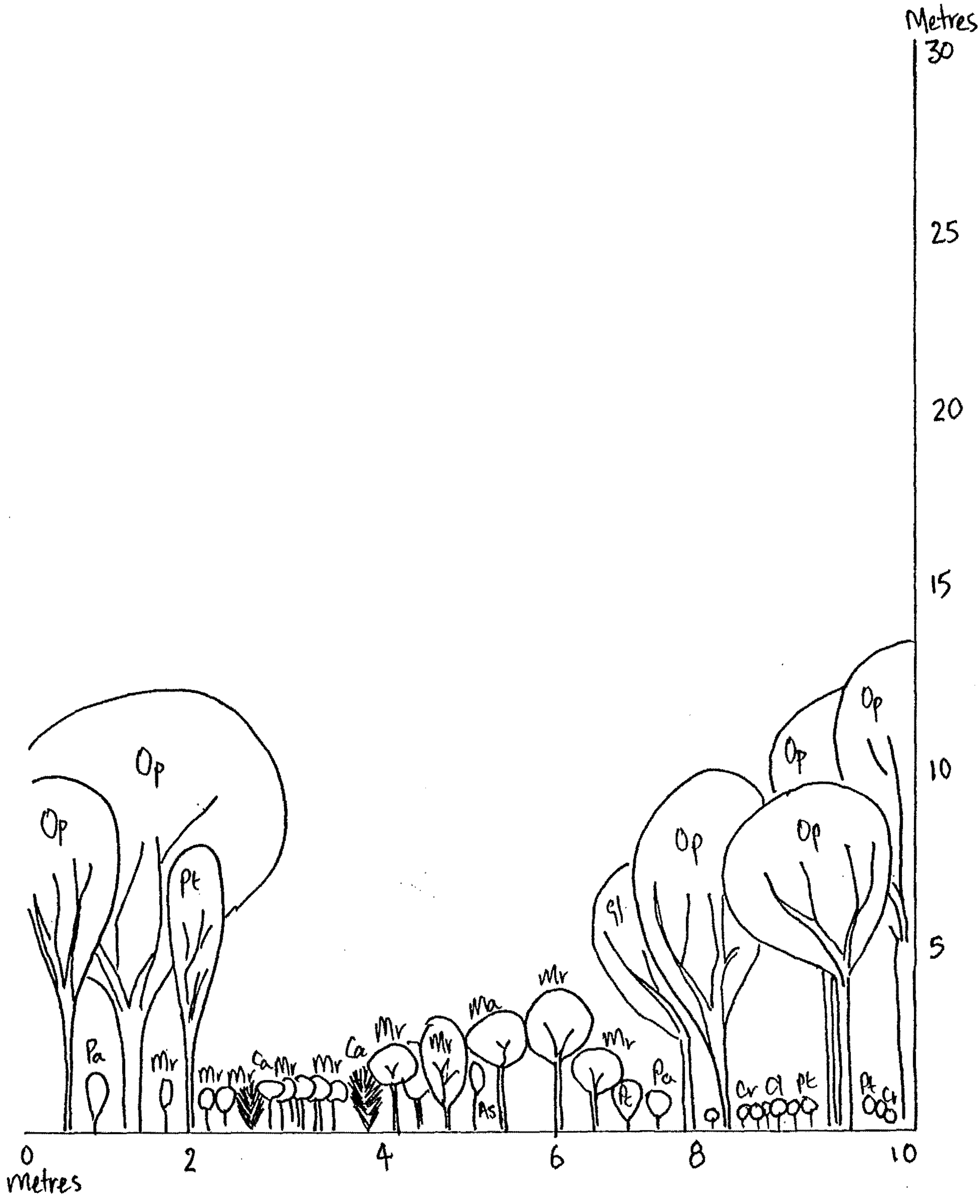
Gl, *Griselinia littoralis*; Ke, *Kunzea ericoides*; Op, *Olearia paniculata*; Pt, *Pittosporum tenuifolium.*

(b) Restoration 2.



Cl, *Coprosma lucida*; Crh, *Coproma rhamnoides*; Gl, *Griselinia littoralis*; Mr, *Melicytus ramiflorus*; Op, *Olearia paniculata*; Pa, *Pseudopanax arboreus*; Pac, *Podocarpus acutifolius*; Pt, *Pittosporum tenuifolium*.

(c) Restoration 3.

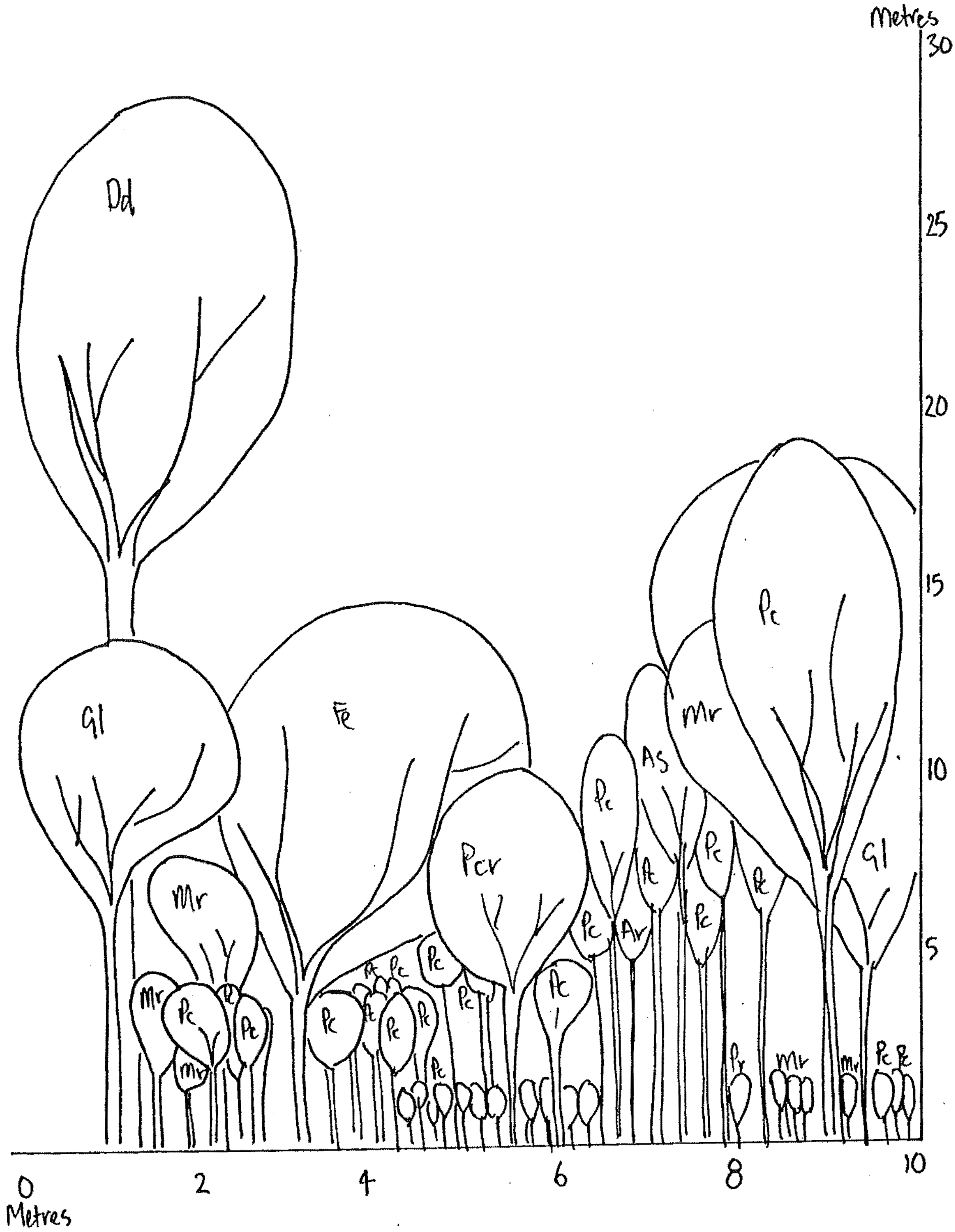


As, *Arsitotelia serrata*; Ca, *Cordyline australis*; Cl, *Coprosma lucida*; Cr, *Coprosma rotundifolia*; Gl, *Griselinia littoralis*; Ma, *Myrsine australis*; Mr, *Melicytus ramiflorus*; Op, *Olearia paniculata*; Pa, *Pseudopanax arboreus*; Pc, *Pseudowintera colorata*; Pe, *Pittosporum eugenoides*; Pt, *Pittosporum tenuifolium*.

(d) Natural Regeneration.

Cr, *Coprosma rotundifolia*; Ke, *Kunzea ericoides*; Ma, *Myrsine australis*; Mr, *Melicytus ramiflorus*; Pa, *Pseudopanax arboreus*; Pc, *Pseudowintera colorata*; Pe, *Pittosporum eugenoides*; Pr, *Plagianthus regius*; Pt, *Pittosporum tenuifolium*.

(e) Mature forest.



As, *Aristotelia serrata*; Dd, *Dacrydium dacrydioides*; Fe, *Fuchsia exorticata*; Gl, *Griselinia littoralis*; Mr, *Melicytus ramiflorus*; Pa, *Pseudopanax arboreus*; Pc, *Pseudowintera colorata*; Pco, *Pennantia corymbosa*; Pcr, *Pseudopannax crassifolius.*

Table 3.2 Basal area of the woody vegetation in the five forested study plots.

DBH (m^2ha^{-1})	AH	NR	R3	R2	R1
Aristotelia serrata	0.59	0	0	0	0
Coprosma linariifolia	0	1.21	0	0	0
Coprosma rotundifolia	1.95	0.27	0	0	0
Cordyline australis	0	0	1.90	0	0
Dacrycarpus dacrydioides	37.20	0	0	0	0
Fuchsia exorticata	13.01	2.90	0	1.94	0
Griselinia littoralis	9.30	0	0.81	0.70	0
Hebe salicifolia	0	0	0.75	0.55	0
Hebe strictissima	0	0	0	0.07	0
Hoheria angustifolia	1.75	2.59	0	0.57	0
Kunzea ericoides	0	19.54	0	0	0.04
Melicope simplex	0.36	0.30	0	0	0
Melicytus ramiflorus	6.30	21.20	0.11	0.20	0
Myrsine australis	0	1.07	0	0	0
Olearia avicenniifolia	0	0	0	1.63	0
Olearia sp.	0	0	3.20	0	0
Oleria paniculata	0	0	32.73	34.22	6.09
Pennantia corymbosa	0	0.30	0	0	0
Pittosporum eugenioides	5.65	7.36	4.49	0.96	0
Pittosporum tennuifolium	0	0	3.18	1.28	0
Plagianthus regius	1.91	5.81	0.13	0	0.07
Podocarpus acutifolius	0	0	0	0.05	0
Podocarpus hallii	0	0	0	0.28	0
Pseudopanax arborus	4.37	0.22	0.06	4.21	0
Pseudopanax crassifolius	0.85	0	0	0	0
Pseudowintera colorata	16.75	0.04	0	0	0
Scheffera digitata	0.09	0.08	0	0	0
Sophora microphyla	0	3.97	0.89	1.46	0.04
Total basal area m^2/ha	100.10	66.86	48.25	48.12	6.24

up to 3 m tall, forming a 50/50 mosaic with the grassland component of the plot. Beneath each tree greater than 1m tall there is a light leaf litter layer with few seedlings.

<u>Restoration 2 (R2: Figure 3.1b, Table 3.2).</u>

The planted tree species in this study plot are approximately 30 years old. *Olearia paniculata* is the dominant small tree. Tree fuchsia *(Fuchsia exorticata)*, 5-finger (*Pseudopanax arboreus*), kowhai, *Pittosporum tenuifolium,* lemonwood (*Pittosporum eugenioides*), broadleaf (*Griselinia littoralis*), *Hebe salicifolia, Hebe strictissima*, mahoe (*Melicytus ramiflorus*) and small leaved *Coprosma species* are all common. Hook grass (*Uncinia sp.*), *Parsonsia sp.*, the ferns *Polystichum richardii, Asplenium bulbiferum* and *Asplenium flacidum* and a few of the species present in the grassland study plot are present as ground cover.

This study plot has a closed canopy approximately 12 metres tall. Beneath the canopy the vegetation is open with few moderate sized shrubs (2-5 m tall). There is a dense litter layer containing many small seedlings (< 50 cm).

<u>Restoration 3 (R3: Figure 3.1c, Table 3.2).</u>

The plantings in this study plot are approximately 35 years old. *Olearia paniculata* is the dominant tree. Other canopy species include lemonwood, kowhai, 5-finger, *Pittosporum tenuifolium*, mahoe, broadleaf and cabbage tree (*Cordyline australis*). Shrub species include ducksfoot (*Pennantia corymbosa*) and small leaved *Coprosma species.* Ground cover vegetation comprises those species listed above for restoration 2.

The tree species in this study plot form a dense canopy approximately 15 m tall. Beneath frequent canopy gaps, shrub species from a dense layer up to 4 m tall. The litter layer is dense and there is a abundance of small seedlings.

Natural regeneration (NR: Figure 3.1d, Table 3.2).

The canopy vegetation of this study plot is approximately 100 years old. This is second-growth kanuka mixed broadleaf forest. Lemonwood, kowhai, 5-finger, mahoe, broadleaf and cabbage tree (*Cordyline australis*) are present in the subcanopy, with a dense shrub layer of small leaved *Coprosma sp.*, *Myrsine australis*, *Melicope simplex* and young individuals of the above angiosperm trees. The ground layer has a well developed covering of the ground cover species listed above, and the ferns *Phymatosorus diversifolius* and *Polystichum vestium*.

Kanuka forms a patchy canopy up to 25 m tall with a subcanopy 5-10 m below. The study plot is relatively open with areas of small seedlings, except beneath canopy gaps where a tall dense shrub layer is present.

Ahuriri Scenic Reserve (AH: Figure 3.1e, Table 3.2).

This is a mixed podocarp angiosperm forest with adult and juvenile matai and adult kahikatea. The canopy layer consists of mahoe, fuchsia, 5-finger and pepperwood. *Coprosma rotundifolia* is abundant in the shrub layer, with *Myrsine australis*, other small leaved *Coprosma species* and regenerating angiosperm species. The undergrowth is thin, although the ferns *Phymatosorus diversifolius*, *Polystichum vestium*, *Asplenium bulbiferum* and *Blechnum discolor* are common. Hookgrass (*Uncinia sp.*), some blackberry and bracken are present. The native climber *Muehlenbeckia australis* is common.

This study plot has a tall (20 m) canopy with a few emergent podocarp trees (up to 30 m). The shrub layer is very dense beneath canopy gaps. Small seedlings are abundant throughout the plot and the ground layer is dominated by fern species

3.3 Field methodology.

3.3.1 General.

Two main groups of organisms were sampled in this study; ground (and litter) invertebrates and vascular vegetation (including trees, shrubs, ferns and herbaceous plants but excluding epiphytic species). Abundance was measured in two different ways depending on the group. The abundance of invertebrates was represented by the number of individuals, while the abundance of vascular vegetation was measured by cover.

Field sampling was based on two intersecting 50 m transects, intercepting at the midpoint of each transect, within each study plot (Figure 3.2). As the study plots are small in area (approximately 0.5-11 ha) this method ensures that a maximum number of samples can be collected while sampling a maximum area of each study plot. Transect length was constrained by the area of the restoration plantings.

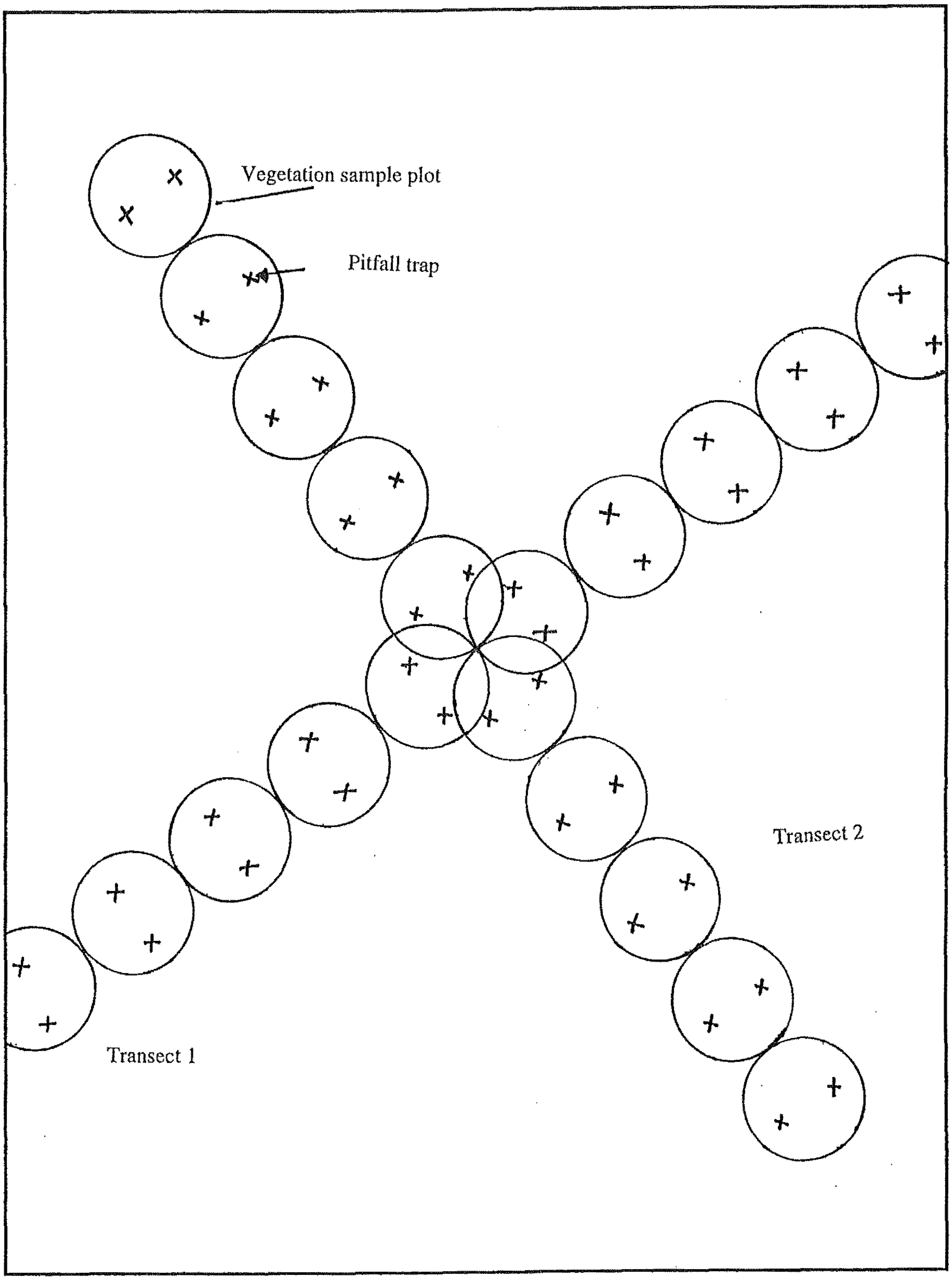
Each transect was laid out along a compass line and was marked with coloured tape for reference throughout the study. Care was taken at all times to avoid trampling or otherwise disturbing the vegetation along each transect.

Sampling was carried out along each transect.

3.3.2 Vascular vegetation.

The vascular vegetation was sampled using a modification of the vegetation recce survey method (Allen and McClennan 1983). This method was originally developed for extensive catchment surveys in montane areas. In the present study, sample plots were circular with a 5 m diameter. The centers of two adjacent sample plots were 5 m apart so that each sample plot abutted the next at a point along the transect (Figure 3.2). The first sample plot was centred 2.5 m from the start of a transect. There were 10 sample

Figure 3.2 Diagram showing the layout of the study plots.



plots per transect, totalling 20 sample plots per study plot. Sample plots were numbered 1-10 along the first transect and 11-20 along the second transect for each study plot. Results from sample plots 15 and 16 for each study plot were discarded due to the overlap with sample plots 5 and 6, thus giving a biased sample for the centre of each study plot as the vegetation here was sampled twice.

Cover within each 5 m diameter sample plot was estimated for each plant species in five strata; ground, shrub, subcanopy, canopy and emergent. Strata heights were <0.5 m, 0.5-3 m, 3-5 m, 5-15 m and 15-25 m respectively for all study plots, with the exception of the naturally regenerating study plot. As this was the only study plot to have subcanopy vegetation, strata heights were changed to give more weight to the subcanopy vegetation layer. The strata heights for naturally regenerating study plot were <0.5 m, 0.5-3 m, 3-7 m, 7-15 and 15-25 m respectively. Cover was estimated using seven cover abundance classes of <1 %, 1-5 %, 6-10 %, 11-25 %, 26-50 %, 51-75 % and 76-100 %. Cover estimates were made from the centre of each sample plot.

With practice this method became relatively simple to use and in good conditions one study plot could be completed within one day. As all measurements were taken by myself and study plots were sampled consecutively, any bias in the results should have been consistent. It is possible that there may be bias with respect to those sample plots first sampled and those sampled last, however I practised my technique before applying it to the sample plots in this study, and was likely to be sampling with reasonable consistency.

Cover is the among most widely used measures of abundance of plant species because it is not biased by the size or distribution of individuals (Floyd & Anderson 1987). Cover-class has been suggested to provide reliable estimates only for dominant shrub species. Estimates for grasses and other species with small or rare individuals are high in comparison with other methods. This is due to the assumption that cover values are uniformly distributed about the mid-points of the cover classes (Floyd & Anderson 1987). This is unlikely to influence results as this study is based on the comparison of the sample plots and biases will be consistent for all sample plots.

Other sources of error include;

(1) Failure to consistently measure all plants in a sample plot in the same way.

(2) Difficulty in determining the vertical boundaries of the sample plots.

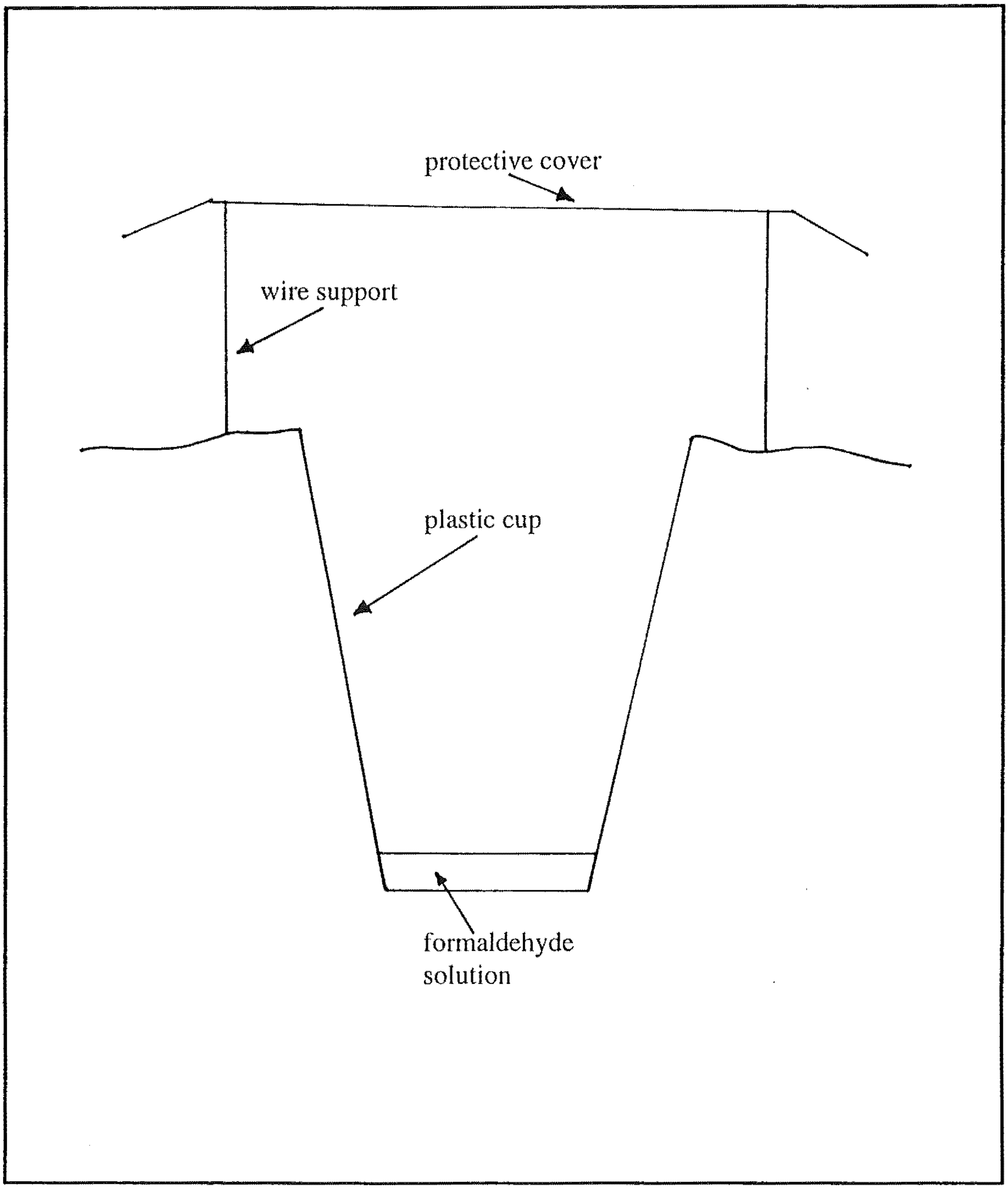
(3) Incorrect identification of some plants.

3.3.3 Ground invertebrates.

All sample plots were sampled for ground invertebrates. Paired pitfall traps were placed 1 m either side of the transects, at 5 m intervals. The first pair of pitfall traps was placed 2.5 m from the start of each transect, thus the pitfall traps were centred within the vegetation sample plot (Figure 3.2). Each transect had 10 pairs of pitfall traps, making 40 pitfall traps for a study plot. As with the vegetation sampling, those pitfall traps placed in the 15 th and 16 th vegetation sample plots were excluded from the analysis. Pitfall trapping is recognised as the most effective method of sampling ground invertebrates with consistency (Thiele 1977). 10-12 pitfall traps were considered sufficient by Walsh (1990) to reveal differences in activity abundance between study plots, and fewer (1-4) traps have been used successfully in other studies of ground invertebrates (eg. Frambs 1990). Pitfall traps are passive, non powered, cheap, easily transported and serviced. They may be left unattended for several days.

Pitfall traps consist of plastic cups placed in the ground so that the tops are level with the surface (Figure 3.3). Care was taken to minimise surface disturbance. At the beginning of each trap period a small amount (20 ml) of 5 % formalin (formaldehyde) solution was placed in each cup. It is recognised that any preserve may have attractant qualities (Adis 1979; Borges 1992), however preservative was essential to stop samples decaying. Care was taken so that the amount of solution was kept constant as this can influence results (Walsh 1990; Holmes et al 1993). Over the cup, a protective cover (a plastic plate on wire supports) was placed leaving sufficient gap to permit the entry of insects, while preventing litter from falling in or birds removing any trapped invertebrates. The cups were removed after 2 weeks of trapping in late December 1994. Invertebrates were trapped from all sample plots at the same time to ensure the data collected from each

Figure 3.3 Diagram of a pitfall trap.



study plot were comparable. Butcher & Emberson (1981) found by pitfall trapping in Ahuriri Scenic Reserve that carabid beetles were most active between November and March. P. Walsh & J. Hutcheson (pers. comm.) suggested that December-January was the best time for a "one off" pitfall assault, as activity abundance was likely to be greatest for the most invertebrates at this time.

In the laboratory, insects were removed from the cups and stored in carefully labelled containers containing 70 % ethanol solution. The insects were sorted on the basis of external morphology to species or taxonomic groups (RTU's), with a representative from each group being identified by Dr P. Walsh or Mr J. Hutcheson, NZ FRI Rotorua. RTU's are taxa that are readily separable by a minimum of unambiguous morphological features. Ambiguity may arise from sexual dimorphism and developmental polymorphism. However, RTU estimates are close to formal taxonomic estimates (Oliver & Beattie 1993). The real advantages of using RTU's are that surveys do not need to be carried out by professional taxonomists and demands on time are greatly reduced.

While pitfall trapping is a technique widely used in ecological studies it has been criticised as being a source of much bias. Sources of bias and error include the following;

(1) Pitfall trapping is biased toward move active mobile species with larger sampling areas. Other more sedentary species may have been under-represented (Halsall & Wratten 1988; Walsh 1990). Pitfall trapping provides a measure of the 'activity' abundance of invertebrate species rather than a measure of the actual populations present (Walsh 1990; Topping & Sunderland 1992; Usher 1992).

(2) Some species may be able to avoid traps or escape from traps once caught (Halsall & Wratten 1988).

(3) Samples may have contained some species which had just wandered into the area and were not characteristic of the invertebrate community of the sample plots.

(4) Many species show temperature and seasonal fluctuation (Greenslade 1964; Speight & Wainhouse 1990).

(4) Ground vegetation and cover may effect invertebrate activity and hence influence trap efficiency (Greenslade 1964; Topping & Sunderland 1992).
(5) There may be differences in activity between sexes (Topping & Sunderland 1992).

While these biases and sources of error may seem considerable they are not likely to greatly influence results as the bias is consistent between study plots.

3.3.4 Environmental data.

The data for the following environmental variables were recorded from the centre of each vegetation sample plot (see methods above) for each study plot. As with the vegetation and invertebrate sampling , the environmental data collected from sample plots 15 and 16 were excluded from the analysis.

Aspect: Aspect was recorded using a SUUNTO hand held compass. The aspect each sample plot was recorded as the direction in which the slope of each study plot faced.

Slope: Slope was recorded using a SUUNTO hand held clinometer. Two 1 m tall poles were vertically positioned opposite each other on the circumference of the 5 m sample plot such that the height difference between the top of the two poles was maximised. The clinometer was placed on top of one of the poles and the top of the opposing pole was sighted. The difference in height of the top of the two poles was recorded on the clinometer as the slope of the sample plot.

Soil moisture: The level of soil moisture was sampled on an overcast day, from the centre of each sample plot by extracting a sample of soil using a soil corer to a depth of 20 cm. Each soil sample was immediately placed in a plastic bag which was then sealed. Samples were clearly labelled. In the laboratory each sample was weighed in its bag before being removed and placed in a metal dish and oven dried at 60°c for 24 hours. The sample was weighed on immediate removal from the oven. The amount of moisture in the soil was calculated as the decrease in soil mass after drying (The mass of the bag included in the initial weighing was accounted for).

Litter Biomass: One litter sample was taken from each sample plot. Litter was collected from a 25 cm x 25 cm quadrat placed in the centre of each sample plot. Stones and large sticks were removed from each sample and the remains were placed in a paper bag and labelled. The paper bags and their contents were oven dried for 24 hours at 60 °c. The samples were then removed from the bags and individually weighed.

Age: The age of the three restoration and the naturally regenerating study plots were estimated by coring 7 of what appeared to be the oldest trees in the naturally regenerating study plot and 5 of what appeared to be the oldest *Olearia paniculata* trees in the restoration study plots. Cores were taken as near as possible to the ground. Aerial photographs taken over a period of 60 years were also used to help estimate the age of each restoration study plot.

Dead wood: The amount of dead wood in each sample plot was estimated on a five point scale, with little or no dead wood scoring 'one' and an abundance of deadwood scoring 'five'.

3.4 Data Analysis.

In order to compare the six study plots the following analyses were used.

The data were divided into the following groups and categories for the analyses.

Group 1 (vascular plants)

- (i) All vascular species
- (ii) Regenerating tree species (only species with canopy potential)
- (iii) Canopy and subcanopy species

Group 2 (invertebrates)

(i) All invertebrate species

(ii) Spiders (including harvestmen)

(iii) Beetles

3.4.1 Vegetation analysis.

Importance values.

Analysis of the vegetation data was based on importance value scores generated from the cover assessment outlined above. Vegetation data were combined to a single importance value (IV) for each species in each sample plot. The IV value was calculated by multiplying the $\log_{10}+1$ of each stratum height by the midpoint of the species cover class, then summing the resulting values for each stratum.

For each of the 3 vegetation categories a new set of IV values was calculated based on the cover of each species in each stratum height. For example the IV scores used for category (i) (all vascular species) were generated using the cover scores of all species present in all strata heights. The IV scores for category (ii) (regenerating tree species) were calculated using the cover scores of the appropriate tree species present as either shrubs or seedlings, as these represent regenerating vegetation. Similarly, the IV scores for the subcanopy and canopy category were based on the cover of those species present only in the canopy and subcanopy.

Diversity indices.

Diversity indices are widely used in ecological studies for comparing different study plots (Magurran 1988; Mathers 1992; Lapin and Barnes 1995; Keesing 1995) and were used in this study for this purpose. Four diversity indices were calculated:

Species Richness (S):

S = number of observed species.

It is not possible to generate an unbiased estimate of true species richness (S) (Smith et al 1979) as there will always be species present in a community which are absent from a sample (Palmer 1990). Palmer found the correlation between the number of species observed (SO) and the true values of species richness (SR) to be 0.973. This value is close to 1, implicating that SO, despite being biased, can be used to compare the SR of different areas provided that the sample size remains constant (Mathers 1992).

Cover: Vegetation cover for each study plot was calculated by summing the importance values (IV) of every species in every sample plot. This was to generate an individual vegetation cover score for each study plot, to enable comparison between each study plot. This was applied to each of the 5 vegetation categories by calculating the IV at the appropriate stratum level.

Diversity (H'): Diversity is a widely used measure in ecological studies and is a valuable tool for comparing different sites. Diversity is based on two components, variety and the relative abundance of species. The diversity of the vegetation within each study plot was calculated using the Shannon Weiner diversity index, or as it has become known, as Shannon's Index (Magurran 1988). It is calculated from the following equation

$$H' = -\sum pi \log_2 pi$$

where *pi* is the proportion of cover represented by the *i*th species.

Evenness (E): Evenness is generally regarded as the measure of equality of abundances in a community (Alatalo 1981). The modified Hill's ratio was applied in this study, calculated using the following equation.

$$E = \frac{(1/\sum pi^{\wedge 2})-1}{[\exp(-\sum pi \ln pi)]-1}$$

where pi is the proportion of cover represented by the ith species.

Modes of dispersal: To assess the role that bird dispersal might play in regeneration, the regenerating vegetation for each study plot was divided into two categories, bird dispersed or not bird dispersed, depending on whether the seed from a particular species was available to birds in the form of food (berry or drupe) based on Burrows (1994a,b).

Floristic similarity: Jaccard coefficients were used as a measure of beta diversity (similarity within each study plot) (Lapin and Barnes 1995). Jaccards coefficient is a binary measure based on presence-absence data.

$$Cj = j/(a+b-j)$$

where j equals the number of species present in both samples, a equals the number of species present in sample 1, and b equals the number of species present in sample 2 (Greig-Smith 1983). One-way analysis of variance (ANOVA) was used to compare statistically the six study plots, using the statistical package SAS. Pairwise multiple comparisons were conducted using Duncans multiple range test to determine the nature of the differences detected by ANOVA.

Similarity index: Jaccards coefficient was used as a measure of similarity to indicate how common the plant species of each study plot were to each other. Jaccards coefficient was used as in the methodology above.

Vertical profile diagrams: Vertical profile diagrams were drawn for each forested study plot to illustrate the vertical structure of the vegetation. All tree species greater than 50 cm in height were mapped along a 10 x 2 m transect. Approximate height, cover and shape of each species was estimated in the field. This three-dimensional information was combined into two-dimensions for the resulting diagrams.

3.4.2 Invertebrate analysis.

For the invertebrate data, recognisable taxonomic units are used instead of species.

Diversity indices.

Species richness (S): Species richness is estimated as with the methods above, with the exception that it is not a true indicator of species richness but RTU richness.

Number of individuals (N): The number of individuals were counted for each study plot, giving a similar representation as calculating the vegetation cover for each study plot.

Diversity (H'): Diversity was calculated using the Shannon's Index as in the vegetation methods above. Diversity was calculated using RTU's instead of species.

Evenness (E): Evenness, as with diversity was calculated as with the vegetation, using the modified Hill ratio, again with RTU's instead of species.

Summed abundance classes: The concept of summed abundance classes was derived by John Hutcheson (pers. comm.). As diversity combines all the data from a study plot into a single number, important information can often be obscured, especially in the case of invertebrates where different individuals play important different roles ecologically.

This technique was applied to the beetle group only, as most ecologically interpretable invertebrate information comes from this group (J. Hutcheson pers. comm, Hutcheson 1990). Each beetle RTU was put into one of three functional categories, based on whether it was a herbivore, predator or detritivore (including algal feeders). The data were transformed from the number of individuals to 5 abundance classes for each RTU in each study plot. The abundance classes were 1, 2-4, 5-9, 10-19, 20+ and were ascribed abundance class values 1 to 5 respectively. The abundance class values are combined for each study plot to give a summed abundance score.

This information is displayed graphically as the number of RTU's in each functional category and as the number of individuals in each category.

Similarity index: Jaccards coefficient was used as a measure of similarity to indicate how common the invertebrates of each study plot were to each other. Jaccards coefficient was used as in the vegetation methodology above.

3.4.3 Ordination Techniques

Ordination techniques organise community data based on species abundances. Environmental interpretation is usually left to a subsequent independent step. All multivariate analyses attempt to summarise community data by producing a low-dimensional ordination space, with similar entities being close together and dissimilar entities far apart (Gauch 1982). Field data must be high dimensional due to the large number of species and samples. On the other hand the final results must be low dimensional to account for human limitations (Gauch 1982). Most ordinations derive an ecological space from the input data. First, ordination summarises community patterns. These patterns are then compared with environmental information to produce an environmental interpretation of the ordination results (Gauch 1982). Ordination may be viewed as the start of the overall process for deriving an ecological space from the input data.

The dominant composition gradients for both the vegetation and the invertebrate and their relationships with the environmental variables were investigated by detrended correspondence analysis (DCA) using the computer program CANOCO (version 2.1; ter Braak 1987). Analysis was undertaken using the default options offered in CANOCO. Invertebrate community data are difficult to analyse and can be "noisy" (P. Walsh pers. comm.). Noisy data can often obscure trends. A possible explanation as to why invertebrate data are noisy may be because invertebrate sampling often collects many uncommon species which are represented by few individuals. For the invertebrate groups, a second ordination was run using the combined invertebrate data from each study plot, as suggested by P. Walsh (pers. comm.) to try and reduce "noise" in the data and clarify patterns.

DCA is an improved eigenvector ordination technique based on Reciprocal Averaging/Correspondence Analysis, but corrects its two main faults (Gauch 1982; Kent & Coker 1992).

The arch distortion effect of RA/CA arises when the second and higher axes are derived, being constrained to be uncorrelated to lower axes. The arch is uncorrelated, yet causes a strong systematic relation of the second axis to the first, which is not wanted (Gauch 1982). The arch is the result of the quadratic relationship between the first and second axes, and is rarely a reflection of the ecological content of the data (Kent & Coker 1992).

The orthogonal criterion for the second and higher axes of RA/CA is replaced in DCA with the criterion that the second and higher axes have no systematic relationships to lower axes. This stronger criterion is implemented by detrending.

Detrending in DCA is a division of the first axis into a number of segments. Within each segment, the second axis scores are adjusted to have an average of zero. Detrending is applied to the sample scores at each iteration. The exception is that once convergence is reached the final sample scores are derived by weighted averages of the species scores without detrending. This results in DCA eigenvector ordination of the species with no arch problem. It also gives a correspondence set of sample scores, simply the weighted average of the species scores, as in RA/CA. To calculate a third DCA axis, sample scores are detrended with respect to the second axis as well as the first, and so on (Gauch 1982).

The other fault of RA/CA is the compression of the ends of the first axis, in relation to the centre of the axis. DCA expands or contracts small segments along the ordination axis so that species turnover occurs uniformly along the species ordination axis. The consequence of this is that equal distances in the ordination correspond to equal differences in species composition (Gauch 1982).

Another important feature of DCA is that the axes are scaled in units of the average standard deviation of species turnover (SD). Along a gradient, a species appears, rises to its mode then disappears, such that a complete turn over in species composition occurs in four standard deviations. A change of 50% in composition (half change) in a sample occurs in one standard deviation. Axes can be of various lengths, unlike RA/CA which are scaled into an arbitrary range of 0-100 depending on the size of the eigenvalues (Kent & Coker 1992). An eigenvalue is associated with each axis. In general the higher the eigenvalue the more significant the axis is at modelling variation in the data. A significant drop from one axis to the next also signifies that the axis associated with the higher value models the dominant cause of variation.

The real value of DCA, however, is in the analysis of difficult data sets. Extensive tests have shown DCA to be as good as RA/CA and nonmetric multidimensional scaling, and usually better than other techniques (Gauch 1982). Because community data are analysed alone, environmental interpretation is a subsequent task. This is aided greatly by the robustness, freedom from distortion and meaningful axis units of DCA.

Like all methods, DCA is not perfect. Outliers and discontinuity both present problems. Outliers are best removed from the data set. Large discontinuity can mean that the widths of gaps in the gradient have to be estimated. This can lead to inaccuracies (Kent & Coker 1992).

Variations in species richness have been noted to distort the RA/CA ordination, in addition to the arch and compression effects. DCA does not necessarily correct for this. Beta diversity has been suggested to be a significant source of distortion, although species richness is also important, as it varies systematically along most successional and environmental gradients (Kent & Coker 1992).

It has been suggested that there is no theoretical justification for DCA. This is because it is an *ad hoc* adjustment of RA/CA and the model is not consistent with the structure of the data. It has been suggested that DCA just flattens the arch, which fails to aid our understanding of the data and does not help to identify the cause of observed distortion.

Also, by flattening the arch, some suggest that there is a loss in ecological information. This is if some of the arch represents a real pattern in the original data (Kent & Coker 1992).

The method of detrending has also been criticised and refined. The original method of subtracting a moving average using segments may be unstable in some cases. Polynomial detrending involves replacing axes scores by residuals from a multiple regression on polynomial functions of the axes already obtained. This provides an alternative for reducing the arch effect but does nothing for reducing the effect of compression (Kent & Coker 1992).

DCA is a widely used indirect ordination technique. Despite some problems, it is as good a technique as most, and better that many in some situations. The interpretation of the results from DCA is best carried out with a knowledge of its limitations and in comparison with other ordinations using the same data (Kent & Coker 1992).

Spearmans rank correlations.

The associations between the first four DCA axis data and environmental variables were tested using the non-parametric technique of Spearmans Rank Correlation, using the statistical package SAS. Comparisons were made between each vegetation and invertebrate category and the environmental variables associated with those categories. While this technique provides is an effective way in which to identify environmental associations with the data, care must be taken when interpreting cause from effects. That is, any relationship identified may not be directly related, but associations may be the consequence of both variables reacting to some other factor.

CHAPTER FOUR

RESULTS and INTERPRETATION- Vegetation

In this chapter the results of the vegetation assessments and ordinations are presented with a primary interpretation.

4.1 Diversity assessments.

Figures 4.1 through 4.3 present graphs of the diversity assessment defined in the previous chapter (species richness, cover, evenness and diversity) against each study plot. Each figure shows the results for one subgroup of organisms.

For most groups diversity (Shannon's Index) and evenness (modified Hill ratio) showed almost identical trends.

4.1.1 Vascular vegetation (Figure 4.1).

Species richness (Figure 4.1a) is lowest in the grassland study plot (27) and highest in restoration 3 (59), with the other two restoration study plots and the naturally regenerating study plot similar (49-59). The mature forest study plot has an intermediate species richness (36). Total vascular plant cover (Figure 4.1b) is the lowest in the grassland study plot and highest at the mature forest study plot with the other study plots intermediate. Evenness (Figure 4.1c) and diversity (Figure 4.1d) show similar patterns between study plots, being lowest in the grassland and restoration 1 study plots, and highest in the mature forest study plot, with the other study plots intermediate.

Figure 4.1 Vascular vegetation. Graphs (a)-(d) show the relationship between study plot and a number of diversity indices (GL= Grassland, R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

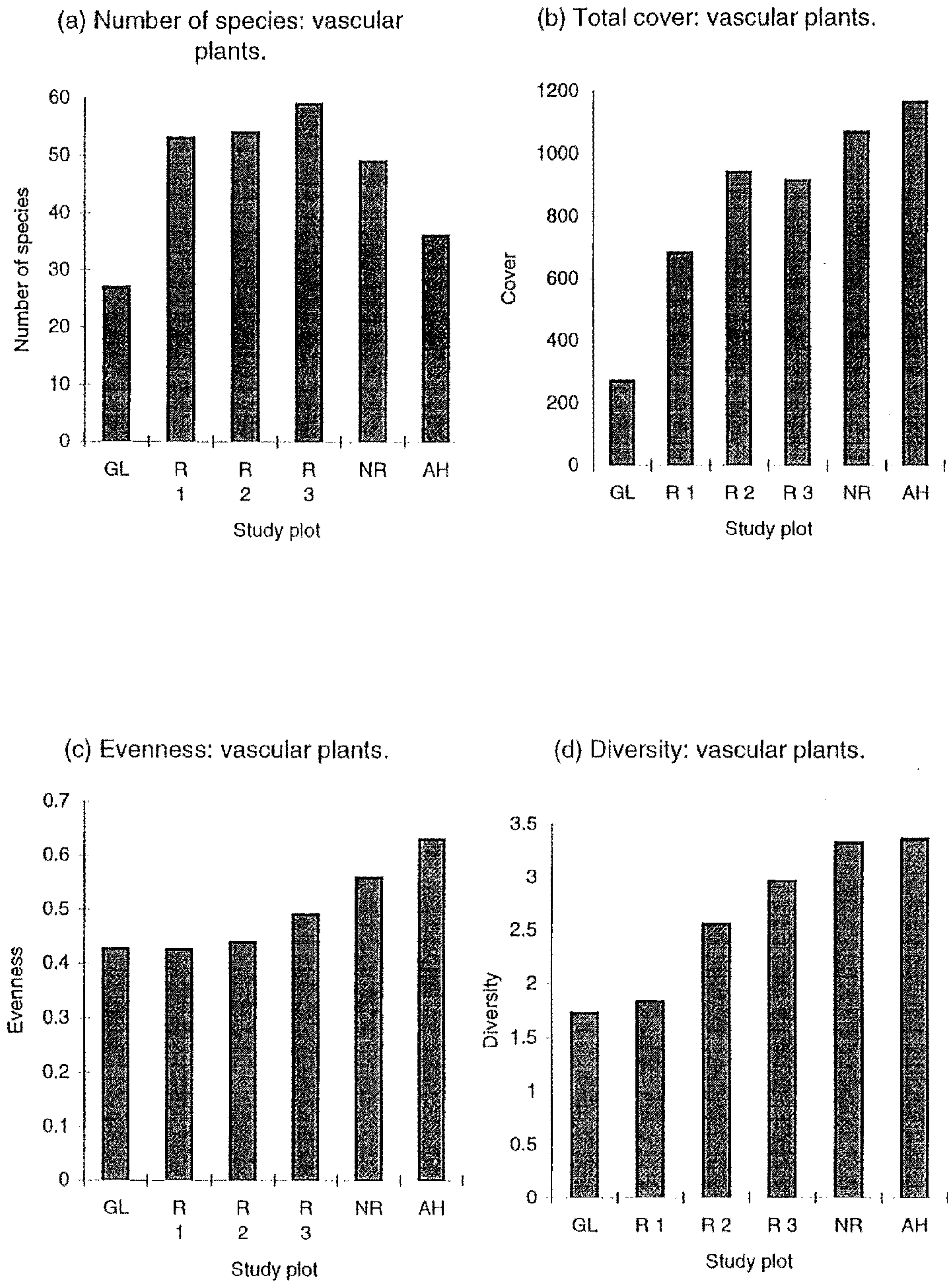


Figure 4.2 Canopy and subcanopy vegetation. Graphs (a)-(d) show the relationship between study plot and a number of diversity indices (R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

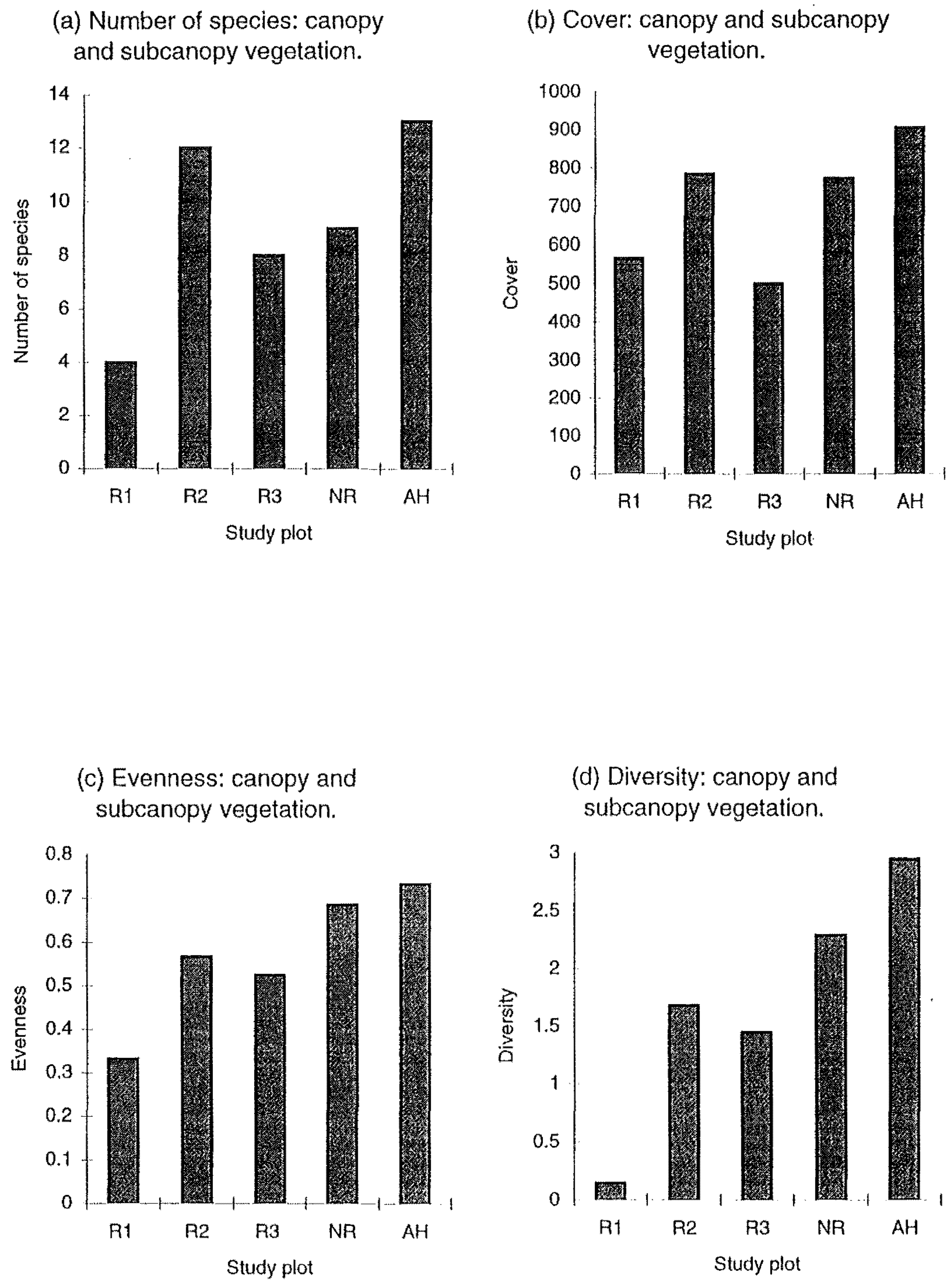
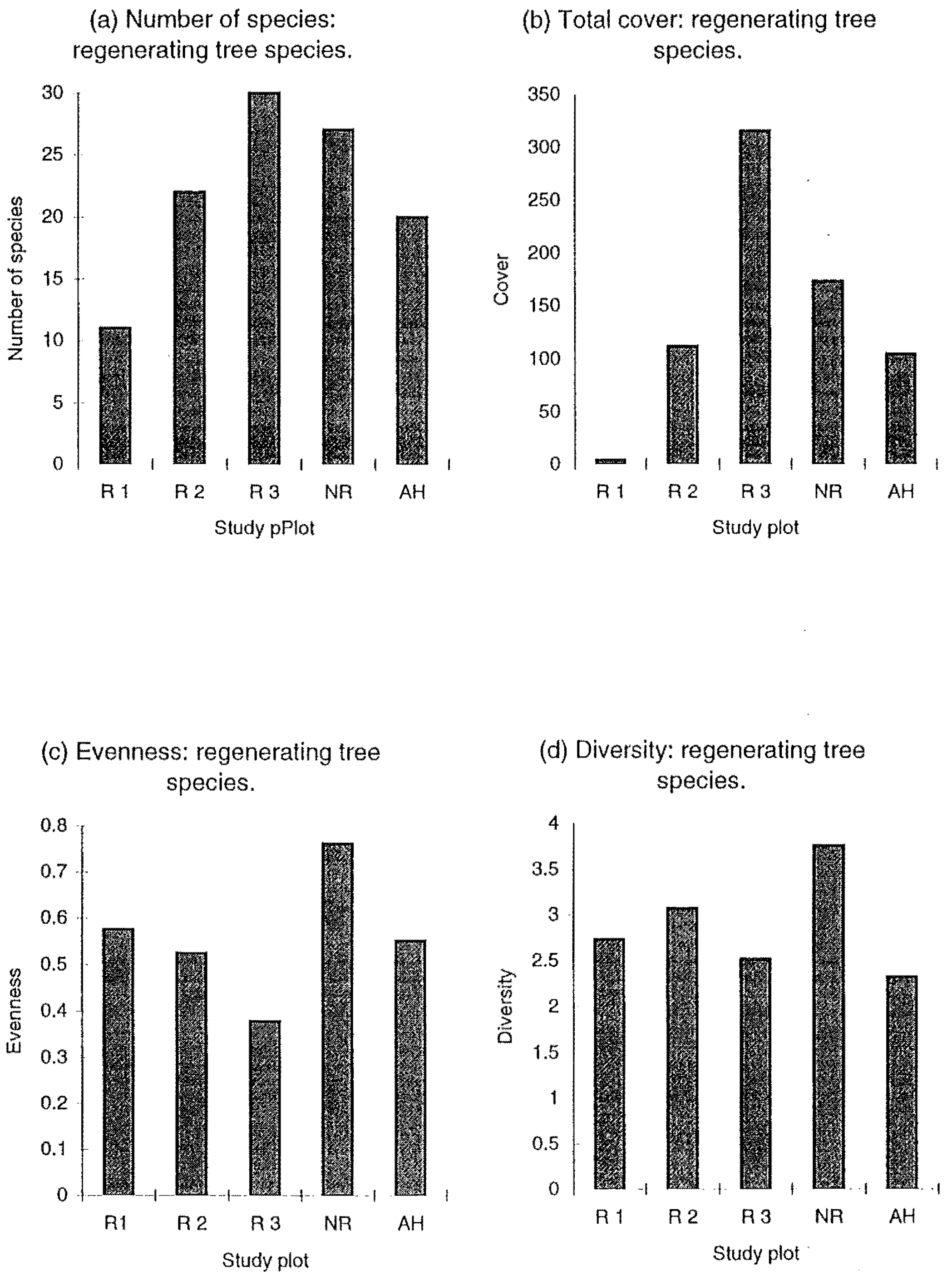


Figure 4.3 Regenerating tree species. Graphs (a)-(d) show the relationship between study plot and a number of diversity indices (R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).



4.1.2 Canopy and subcanopy vegetation (Figure 4.2).

The grassland study plot is not included in this analysis as there are no canopy and subcanopy species present.

Species richness (Figure 4.2a) is lowest in restoration 1 (4) and highest in the mature forest study plot (13) and restoration 2 (12), with the other two study plots similar. Total canopy and subcanopy cover (Figure 4.2b) is lowest in restoration 3 and restoration 1, and highest in the mature forest study plot, with the restoration 2 and the naturally regenerating study plot intermediate. Evenness (Figure 4.2c) and diversity (Figure 4.2d) show similar patterns between study plots, being lowest in restoration 1, and highest in the mature forest study plot. The other two restoration study plots and the naturally regenerating study plot are intermediate.

4.1.3 Regenerating vegetation (Figure 4.3).

The grassland study plot is not included in this analysis as there are no regenerating tree species present.

Species richness (Figure 4.2a) is lowest in restoration 1 (11) and highest in restoration 3 (30) and the naturally regenerating study plot (27), with restoration 2 and the mature forest study plot similar. Regeneration cover (Figure 4.3b) is lowest in restoration 1 and highest in restoration 3. Restoration 2 and the mature forest study plots have similar regeneration cover, with the regeneration cover of the naturally regenerating study plot being slightly higher. Evenness (Figure 4.3c) is highest in restoration 1 and lowest in restoration 3. Restoration 2 and the mature forest study plot have similar evenness, the evenness of the naturally regenerating study plot being slightly higher. Diversity (Figure 4.3d) is lowest in the mature forest study plot and highest in the naturally regenerating study plot, and similar for the remaining three study plots.

4.2 Dispersal mechanisms (Figure 4.4, Table 4.1).

Of the 31 regenerating tree species found in the six study plots, 23 are primarily dispersed by avian dispersal agents. Of the remaining nine, six are wind dispersed and three are dispersed by gravity (Table 4.1).

Most of the regenerating tree species (Figure 4.4a) in each study plot have bird dispersed fruit (64-90%) with fewer species having wind dispersed fruit (10-27%) or having fruit with no obvious dispersal mechanism (9-11%). The mature forest study plot lacked any species with wind dispersed fruit.

Regenerating species with bird dispersed fruit account for the greatest proportion (85-96%) of cover in all study plots, with the exception of restoration 1 where bird dispersed regenerating species account for only 42% of the regeneration cover (Figure 4.4b). The remaining regeneration cover of restoration 1 is wind dispersed species (51%), with 7% being dispersed by gravity. Gravity dispersed species account for 4-7% of the regeneration cover in the study plots. Wind dispersed species account for 9% and 3% of the respective regeneration cover of restoration 2 and restoration 3. The naturally regenerating study plot has less than 1% wind dispersed regeneration, while the mature forest study plot lacks wind dispersed regeneration.

4.3 Floristic similarity (Table 4.2).

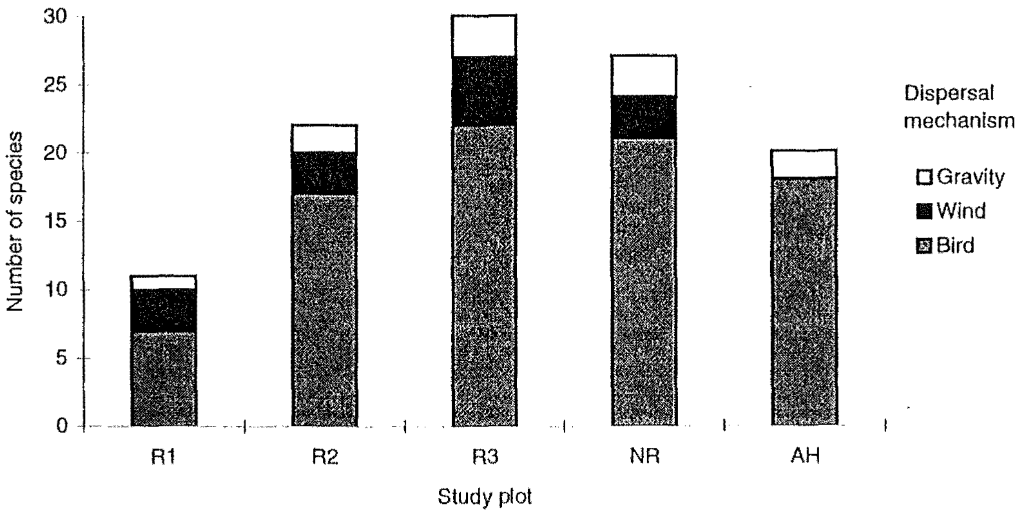
The within study plot floristic similarity was significantly different between the grassland and restoration 1 study plots, then in the restoration 2, restoration 3, natural regeneration and mature forest study plots ($F=23.86$, $P<0.001$) (Table 4.2).

Table 4.1 Dispersal mechanisms of the regenerating tree species.

Regenerating tree species	**Dispersal mechanism**
Aristotelia serrata	Bird
Carpodetus serratus	Bird
Coprosma linarifolia	Bird
Coprosma lucida	Bird
Coprosma propinqua	Bird
Coprosa robusta	Bird
Coprosma rotundifolia	Bird
Cordyline australis	Bird
Dacrycarpus dacrydioides	Bird
Fuchsia exorticata	Bird
Griselinia lttoralis	Bird
Melicytus ramiflorus	Bird
Myrsine australis	Bird
Pennantia corymbosa	Bird
Pittosporum eugenoides	Bird
Pittosporum tenuifolium	Bird
Podocarpus hallii	Bird
Prumnoptiys taxifolia	Bird
Pseudopanax arboreus	Bird
Pseuodpanax crassifolius	Bird
Pseudowintera colorata	Bird
Scheflera digitata	Bird
Hebe salicifolia	Wind
Hebe strictissima	Wind
Hoheria angustifolia	Wind
Kunzea ericoides	Wind
Olearia avicenniifolia	Wind
Olearia paniculata	Wind
Melicope simplex	Gravity
Plagianthus regius	Gravity
Sophora microphylla	Gravity

Figure 4.4 Dispersal mechanisms at the three restoration, naturally regenerating and mature forest study plots (R1= Restoration 1, R2= restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

(a) Number of regenerating tree species per site and their dispersal mechanism.



(b) Proportion of regenerating tree species (based on cover abundance) for each dispersal agent.

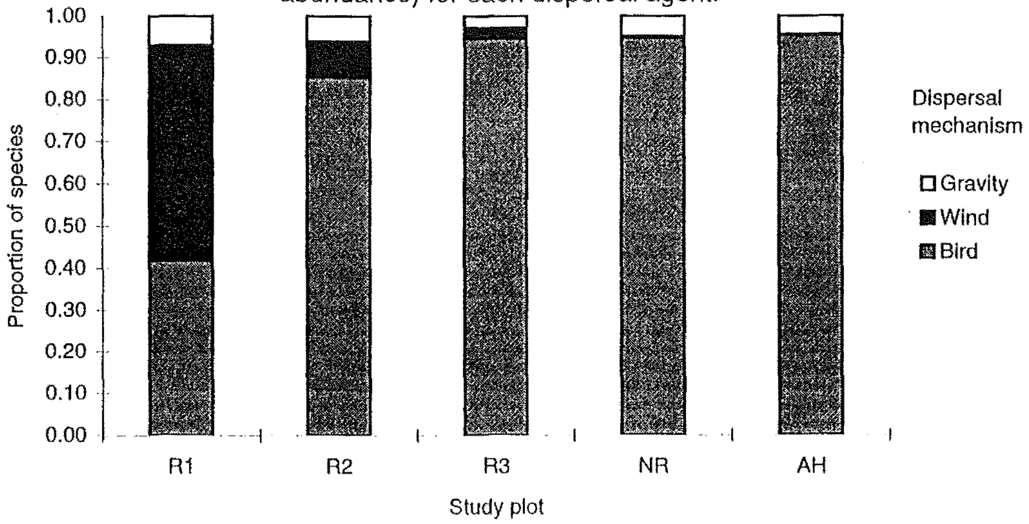


Table 4.2 Mean floristic similarity within study plots (means with the same letter are not significantly different).

Plot	Mean	Standard Deviation
GL	0.41 a	0.430
R1	0.36 a	0.440
R2	0.19 b	0.278
R3	0.14 b	0.210
AH	0.14 b	0.219
NR	0.13 b	0.209

GL= Grassland study plot
R1= Restoration 1 study plot
R2= Restoration 2 study plot
R3= Restoration 3 study plot
NR= Naturally regenerating study plot
AH= Mature forest study plot

Table 4.3 Floristic similarity (Jaccards coefficient) between study plots.

	GL	R1	R2	R3	NR	AH
GL	1.00					
R1	0.37	1.00				
R2	0.56	0.33	1.00			
R3	0.84	0.40	0.71	1.00		
NR	0.52	0.24	0.20	0.69	1.00	
AH	0.54	0.21	0.70	0.25	0.60	1.00

GL= Grassland study plot
R1= Restoration 1 study plot
R2= Restoration 2 study plot
R3= Restoration 3 study plot
NR= Naturally regenerating study plot
AH= Mature forest study plot

4.4 Number of species in common (Table 4.3).

The plant species in restoration 2 and the naturally regenerating study plots are least similar to each other (Jaccards coefficient= 0.20) while the plant species in restoration 3 and the grassland study plots are most similar (Jaccards coefficient= 0.84) (Table 4.3). Restoration 1 is most similar to restoration 2 & 3, restoration 2 is most similar to restoration 3 and the mature forest study plot and restoration 3 is most similar to the grassland and naturally regenerating study plots. The naturally regenerating study plot is most similar to restoration 3 and the mature forest study plot.

4.5 Ordination.

4.5.1 Vascular vegetation (Figure 4.5, Table 4.4).

The DCA ordination (Figure 4.5, Table 4.4a) based on vascular vegetation cover shows a trend across the 1st axis of the ordination scatter from grassland to naturally mature forest study plots, with the restoration study plots ordered from youngest to oldest between these. The grassland, restoration 1 and restoration 2 study plots show little variation along axis 2. The restoration 3 study plots show more axis 2 variation, the naturally regenerating and mature forest study plots having the greatest spread along the 2nd axis. The first axis (eigenvalue= 0.844) accounts for 11.9 % of the variation in the data, while axis 2 (eigenvalue= 0.589) accounts for 8.3 % of the variation in the data. The gradient length of the first axis is 5.60 and the 2nd axis gradient length is 3.55.

Spearman rank correlation coefficients (Table 4.4b) show a high correlation between the 1st vegetation axis and study plot age (0.921) and a moderate correlation between the 1st vegetation axis and aspect (0.663). There were no significant correlations between the 1st vegetation axis and slope or moisture and no significant correlations between the 2nd vegetation axis and any of the environmental variables.

Figure 4.5 DCA ordination of vascular vegetation. The correlation of the most important criteria are shown (length of line is proportional to correlation size) (GL= Grassland, R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

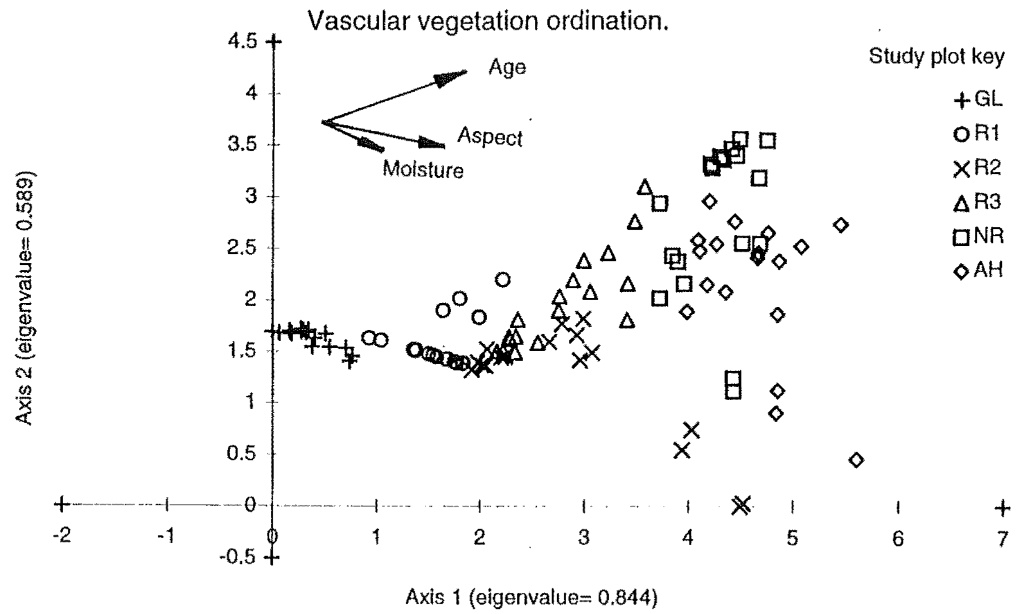


Figure 4.6 DCA ordination of canopy and subcanopy vegetation. The correlation of the most important criteria are shown (length of line is proportional to correlation size) (R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

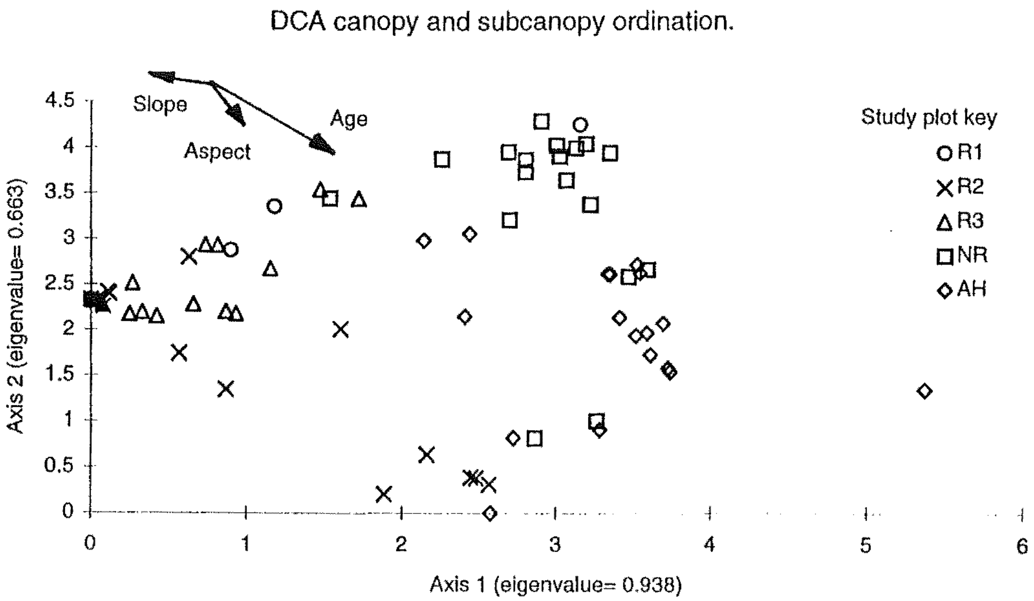


Figure 4.7a DCA ordination of regenerating vegetation (No significant correlations) (R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

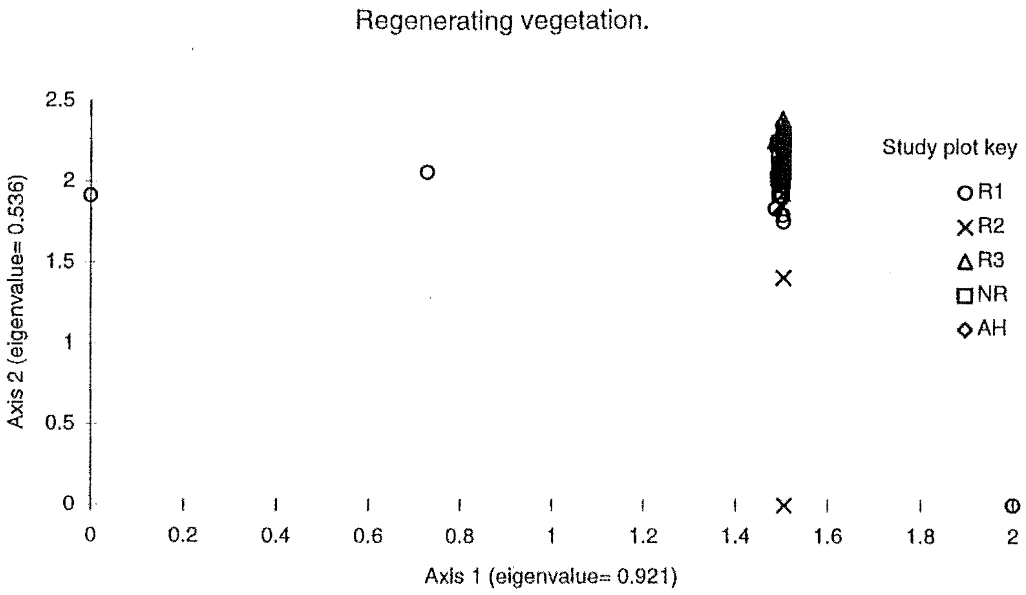


Figure 4.7b DCA ordination of regenerating vegetation (sample plots R1 1, R1 2 and R2 14 omitted) (No significant correlations) (R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

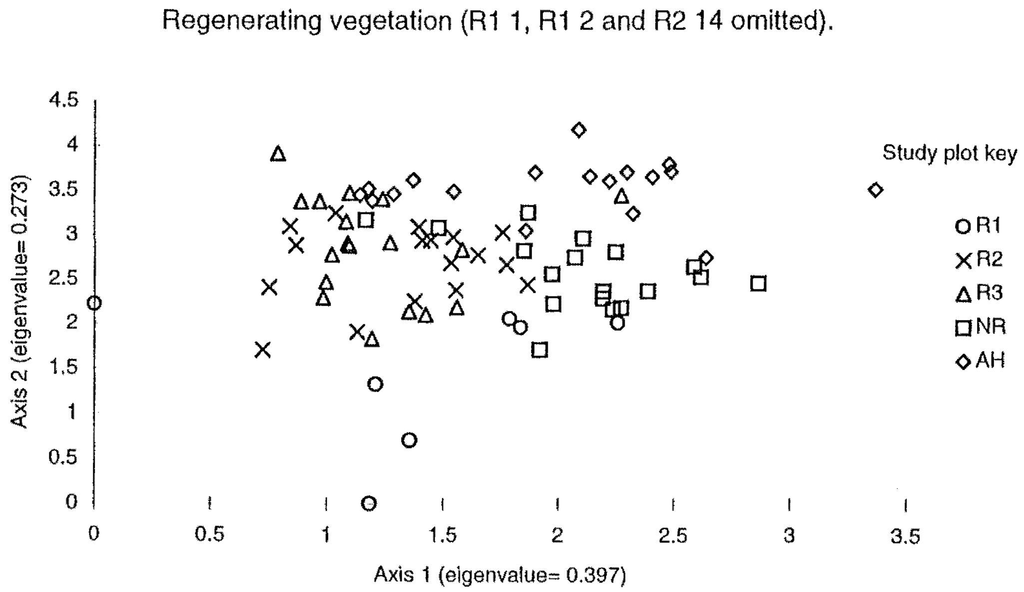


Figure 4.8a DCA ordination of the restoration 3 canopy and subcanopy vegetation, with regenerating vegetation.

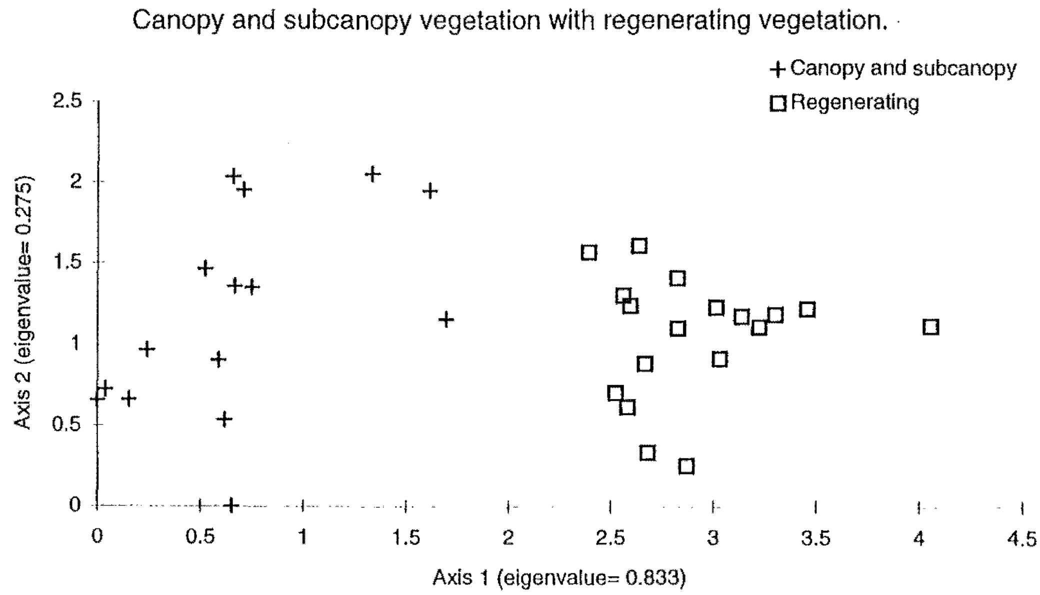


Figure 4.8b DCA ordination of the natural regeneration canopy and subcanopy vegetation, with regenerating vegetation.

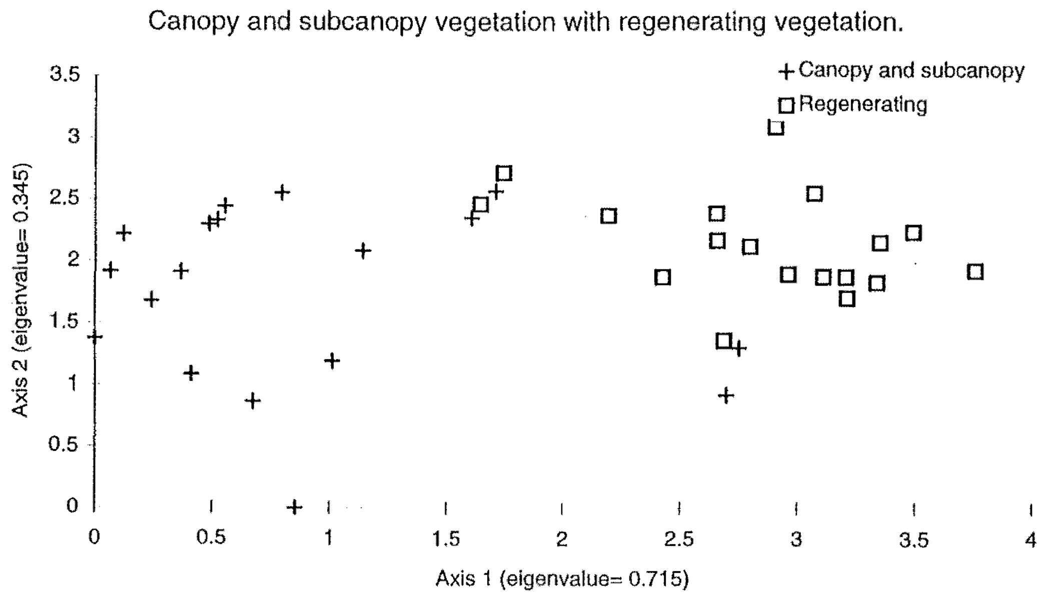


Figure 4.8c DCA ordination of the mature forest canopy and subcanopy vegetation, with regenerationg vegetation.

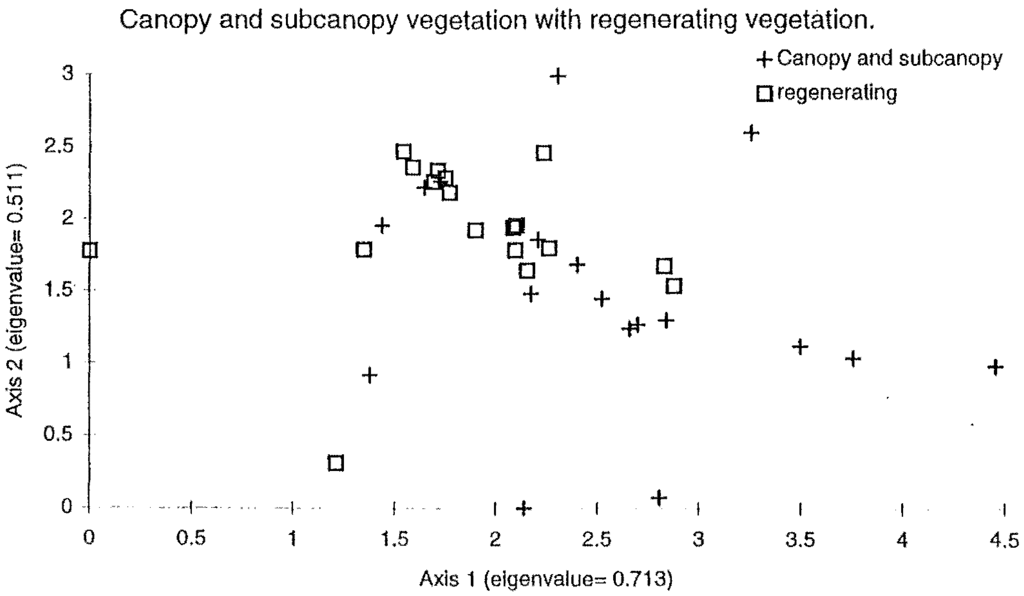


Table 4.4a Vegetation DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.844	0.589	0.283	0.235
Lengths of gradient	5.603	3.553	2.718	2.687
Cumulative percentage variance of species data	11.9	20.2	24.2	27.5
Total sum of eigenvalues	7.090			

Table 4.4b Spearman rank correlation coefficients calculated between the environmental variables and the first two DCA vegetation ordination axes.

	Aspect	Slope	Moisture	Age
Axis 1	0.663	-0.003	0.436	0.921
Axis 2	0.143	-0.236	0.084	0.480

Table 4.5a Canopy and Subcanopy vegetation DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.938	0.663	0.435	0.289
Lengths of gradient	5.371	4.283	3.733	2.255
Cumulative percentage variance of species data	12.9	22.0	27.9	21.9
Total sum of eigenvalues	7.292			

Table 4.5b Spearman rank correlation coefficients calculated between the environmental variables and the first two DCA canopy and subcanopy vegetation ordination axes.

	Aspect	Slope	Moisture	Age
Axis 1	0.611	-0.592	0.190	0.804
Axis 2	-0.300	-0.075	-0.292	0.029

Table 4.6a Regenerating vegetation DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.921	0.536	0.379	0.224
Lengths of gradient	1.506	.386	3.686	3.260
Cumulative percentage variance of species data	20.5	32.4	40.8	45.8
Total sum of eigenvalues	4.498			

Table 4.6b Regenerating vegetation (plots R1 1, R1 2 and R2 14 omitted) DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.397	0.273	0.167	0.142
Lengths of gradient	3.367	4.182	2.538	2.691
Cumulative percentage variance of species data	12.5	21.1	26.4	30.9
Total sum of eigenvalues	3.171			

Table 4.6c Spearman rank correlation coefficients calculated between the environmental variables and the first two DCA regenerating vegetation ordination axes.

	Aspect	Slope	Moisture	Age
Axis 1	-0.047	0.403	0.472	0.356
Axis 2	0.000	0.000	0.000	0.000

Table 4.6d Spearman rank correlation coefficients calculated between the environmental variables and the first two DCA regenerating vegetation (plots R1 1, R1 2 and R2 14 omitted) ordination axes.

	Aspect	Slope	Moisture	Age
Axis 1	0.217	-0.439	-0.270	0.493
Axis 2	0.565	0.387	0.387	0.574

Table 4.7a Restoration 3 (canopy and subcanopy with regenerating vegetation) DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.833	0.275	0.143	0.079
Lengths of gradient	4.056	2.053	2.037	1.373
Cumulative percentage variance of species data	27.1	36.0	40.6	43.2
Total sum of eigenvalues	3.081			

Table 4.7b Natural regeneration (canopy and subcanopy with regenerating vegetation) DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.715	0.345	0.209	0.136
Lengths of gradient	3.758	3.085	2.386	1.989
Cumulative percentage variance of species data	19.6	29.1	34.9	38.6
Total sum of eigenvalues	3.641			

Table 4.7c Mature forest (canopy and subcanopy with regenerating vegetation) DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.713	0.511	0.138	0.078
Lengths of gradient	4.455	2.988	1.911	2.535
Cumulative percentage variance of species data	16.5	28.3	31.5	33.3
Total sum of eigenvalues	4.328			

4.5.2 Canopy and subcanopy vegetation (Figure 4.6, Table 4.5).

The DCA ordination (Figure 4.6, Table 4.5a) separates the five study plots based on canopy and subcanopy vegetation cover. Axis 1 separates the three restoration study plots from the naturally regenerating and mature forest study plots. Axis 1 (eigenvalue= 0.938) explains 12.9 % of the variation in the data. Axis 2 (eigenvalue= 0.663) explains 9.1 % of the variation in the data. The length of the 1st and 2nd axes gradients are 5.37 and 4.28 respectively.

Spearman rank correlation coefficients (Table 4.5b) show a high correlation between study plot age and the 1st canopy and subcanopy vegetation axis. Moderate and weak correlations exist between aspect (0.611) and slope (-0.592) and the 1st axis respectively. No significant correlations exist between moisture and the 1st axis, or the 2nd axis and all the environmental variables.

4.5.3 Regenerating vegetation (Figure 4.7, Table 4.6).

The DCA ordination (Figure 4.7a, Table 4.6a) does not show any trends with respect to the regenerating vegetation of these study plots. The first axis (eigenvalue= 0.921) explains 20.5 % of the variation in the data. Axis 2 (eigenvalue= 0.536) explains a further 11.9 % of the variation. The gradient lengths of the 1st and 2nd axes are 1.51 and 2.39 respectively. The study plots cover 2 standard deviations on the 1st axis, hence there is much similarity of species within each study plot.

Spearman rank correlation coefficients (Table 4.6c) show no significant correlations between the 1st and 2nd regenerating vegetation axes and the environmental variables.

The DCA ordination (Figure 4.7b, Table 4.6b) with outlier sample plots R1 1, R1 2 and R2 14 omitted shows only weak separation between the study plots. The first axis (eigenvalue= 0.397) explains 12.5 % of the variation in the data. Axis 2 (eigenvalue=

0.273) explains a further 8.6 % of the variation. The gradient lengths of the 1st and 2nd axes are 3.37 and 4.18 respectively.

Spearman rank correlation coefficients (Table 4.6d) show no significant correlations between the 1st regenerating vegetation axis (outlier sample plots R1 1, R1 2 and R2 14 omitted) and all the environmental variables measured. Slope and moisture are also not significantly correlated with the 2nd axis. The 2nd axis is weakly correlated with aspect (0.565) and study plot age (0.574).

4.5.4 Canopy and subcanopy and regenerating vegetation (Figure 4.8, Table 4.7).

These ordinations indicate the floristic similarity of the regenerating vegetation to the canopy and subcanopy vegetation of the same study plot, for restoration 3, the natural regeneration and the mature forest study plots, to provide an indication of the similarity of the future canopy to the present canopy.

1. Restoration 3 (Figure 4.8a, Table 4.7a).

The DCA ordination (Figure 4.8a, Table 4.7a) shows a clear separation between the canopy and subcanopy vegetation and the regenerating vegetation of restoration 3. The first axis (eigenvalue= 0.833) explains 27.1 % of the variation in the data and axis 2 (eigenvalue= 0.275) explains a further 8.9 % of the variation. The gradient lengths of the 1st and 2nd axes are 4.06 and 2.05 respectively.

2. Natural regeneration (Figure 4.8b, Table 4.7b).

The DCA ordination (Figure 4.8b, Table 4.7b) shows a clear separation between canopy and subcanopy vegetation and the regenerating vegetation of the naturally regenerating study plot. The first axis (eigenvalue= 0.715) explains 19.6 % of the variation in the data. Axis 2 (eigenvalue= 0.345) explains a further 9.5 % of the variation. The gradient lengths of the 1st and 2nd axes are 3.76 and 3.09 respectively.

3. Mature forest (Figure 4.8c, Table 4.7c).

The DCA ordination (Figure 4.8c, Table 4.7c) does not show a separation of canopy and subcanopy vegetation and regenerating vegetation on either of the first two axes. The first axis (eigenvalue= 0.713) explains 16.5 % of the variation in the data. Axis 2 (eigenvalue= 0.511) explains a further 11.8 % of the variation. The gradient lengths of the 1st and 2nd axes are 4.46 and 2.99 respectively.

4.6 Primary interpretation of vegetation results.

The overall picture from the vegetation analysis is the chronological ordering of the study plots, suggesting a sequence of floristic development associated with study plot age.

4.6.1 Canopy and subcanopy vegetation.

Both the ordination (Figure 4.6) and diversity assessments (Figure 4.2) of the canopy and subcanopy vegetation of the five forested study plots illustrate the floristic differences between the three restoration study plots and the naturally regenerating and mature forest study plots.

The ordination of the canopy and subcanopy vegetation (Figure 4.6) clearly shows this separation of study plots along the 1st axis. While highly correlated with study plot age (Table 4.5b) this separation of plots is likely to reflect the dominant planted component of the three restoration study plots. Therefore, the relative positions of the canopy and subcanopy vegetation of the three restoration study plots on the ordination axes indicates that the species used in the initial plantings in these study plots, particularly *Olearia paniculata*, are not similar to the canopy vegetation of the remnant vegetation of the area.

The diversity assessments (Figure 4.2) also illustrate these differences in the canopy and subcanopy vegetation of the study plots. Both the evenness and diversity of the three restoration study plots are less than the naturally regenerating and mature forest plots (Figure 4.2c, 4.2d) reflecting the more simple planted nature of the restoration study plots. The low species richness and diversity (Figure 4.2a, 4.2d) of the restoration 1 study plot may reflect a lack of species diversity used in the initial planting. Restoration 2 & 3 appear to have been planted using a greater diversity of species (Figure 4.2d, 4.2a). The higher, total canopy and subcanopy cover (Figure 4.2b) in restoration 2 may reflect a greater planting density. However, the lower canopy and subcanopy cover in restoration 3 may relate to windthrow and other gap creating disturbances rather than to initial planting density.

Summary.

The results of the canopy and subcanopy vegetation assessments and ordination reflect the different canopy and subcanopy compositions of the three restoration study plots compared with the canopy and subcanopy of the naturally regenerating and mature forest study plots. This difference reflects differences between the planted nature of the three restoration study plots to the naturally regenerating and mature forest study plots which arose through natural ecological processes.

4.6.2 Regenerating vegetation.

Both the ordinations (Figure 4.7) and diversity assessments (Figure 4.3) of the regenerating vegetation of the five forested study plots illustrate the floristic similarities of the five study plots.

The ordination scatter of the regenerating vegetation (Figure 4.7a) show no obvious separation of the study plots. However, removal of outlying sample points (Figure 4.7b) showed a slight separation with respect to the regeneration of the grassland and mature forest study plots on the 2nd ordination axis. However, the overwhelming impression from these ordinations is that despite canopy and subcanopy vegetation differences, the regenerating vegetation is very similar between the five study plots. This suggests that in this study regenerating vegetation is not greatly influenced by the canopy and subcanopy vegetation of the study plots. Furthermore, the regenerating vegetation of the study plots does not appear to show an ecologically significant correlation with any of the measured environmental variables (Table 4.6c,d).

The regenerating vegetation diversity assessments (Figure 4.3) illustrate the greater cover (Figure 4.3b) and species richness (Figure 4.3a) of the restoration 3 study plot compared with the other study plots. It is possible that this is a consequence of the greater abundance of canopy gaps, providing increased opportunities for the establishment and growth of new individuals.

The majority of regenerating tree species are bird dispersed (Table 4.1). The proportion of bird dispersed regeneration increases from the restoration 1 to mature forest study plots (Figure 4.4b). This suggests that as the study plots age they are becoming more attractive to those bird species which disperse seed. This is highlighted by the relative absence of bird dispersed regeneration from restoration 1, indicating that this study plot has not developed those attributes required to attract birds.

The ordination diagrams of the canopy and subcanopy vegetation with regenerating vegetation (Figure 4.8) of the restoration 3, naturally regenerating and mature forest study plots illustrate the floristic similarity of these strata. In interpreting these ordinations it is possible to obtain an indication of the potential future canopy of each study plot based on the assumption that those tree species regenerating in each study plot are likely candidates for future canopy recruitment. Interpretations based on this assumption indicate that the regenerating vegetation of the mature forest study plot is most similar to its canopy and that this study plot is likely to be in a state of self perpetuation. As the regeneration of the other forested study plots is floristically similar to that in the mature forest study plot, it would seem likely that these plots will develop toward a canopy similar to the mature forest study plot.

Summary.

The regenerating vegetation of the study plots does not appear to be strongly correlated with canopy and subcanopy vegetation or any measured environmental factor. As a high proportion of regeneration is bird dispersed it appears that those factors influencing the regeneration of a study plot are associated with a study plot's potential to attract birds. The current regeneration patterns in the study plots and the similarity of regeneration floristics to canopy and subcanopy floristics indicates the future canopy of the three restoration and naturally regenerating study plots will resemble the current canopy of the mature forest study plot.

4.6.3 Total vascular vegetation.

Both the ordination (Figure 4.5) and diversity assessments (Figure 4.1) of the vascular vegetation of the six study plots illustrate the floristic similarities of the study plots to each other.

The vascular vegetation diversity assessments (Figure 4.1) suggest an overall developmental trend from grassland to mature forest study plots. While the total number of vascular plant species is highest in restoration 3 (Figure 4.1a) this is likely to reflect a community where both grassland and forest species are present. Overall species richness should decline to levels similar to the naturally regenerating and mature forest plots when the grassland component is lost. The total cover, evenness and diversity all show an increasing trend with study plot age (Figure 4.1a,b,c) illustrating the developmental trend of the study plots from a species poor, low diversity community to a more species rich, diverse community.

The ordination of vascular vegetation (Figure 4.5) again illustrates this developmental trend. As the study plots age they become more floristically similar to the mature forest study plot. The strong correlation of these axes with age (Table 4.4b) supports this. The increasing scatter of sample plots on the 2nd axis with increasing plot age illustrates the development of floristic variation within the study plots. The position and scatter of the sample plots of restoration 3 on the ordination scatter may be interpreted as an indication that this is the first restoration study plot to develop characteristics similar to the naturally regenerating and mature forest study plots. The grassland, restoration 1 and restoration 2 study plots are relatively clumped with respect to the 2nd axis in comparison with the remaining plots. The length of the 1st axis indicates that few species present in the grassland study plot are present in the mature forest study plot and vice versa.

The mean floristic similarity values (Table 4.2) group the grassland and restoration 1 plots together and the restoration 2, restoration 3, the naturally regenerating and mature forest plots. This indicates that the restoration 1 plot has failed to develop the compositional heterogeneity of the other restoration and naturally regenerating and mature forest plots and suggests that this study plot strongly retains those traits characteristic of a planting.

The similarity of the plant species in each study plot to each other (Table 4.3) shows a floristic progression from restoration 1 to the naturally regenerating study plot.

<u>Summary.</u>

The results of the vascular vegetation assessments illustrate the overall developmental trend of the study plots toward the mature forest study plot with increasing study plot age.

CHAPTER FIVE

RESULTS and INTERPRETATION-invertebrates

In this chapter the results of the invertebrate assessments are presented, with a primary interpretation for the invertebrates.

5.1 Diversity assessments.

Figures 5.1 through 5.3 present graphs of the diversity assessment defined in the previous chapter 3 (species richness, number of individuals, evenness and diversity) against each study plot. Each figure shows the results for one subgroup of organisms.

5.1.1 Invertebrates (Figure 5.1).

Species richness (Figure 5.1a) is lowest in the grassland study plot (39) and highest in the mature forest study plot (59), with the remaining plots similar. The total number of individuals (Figure 5.1b) is lowest in restoration 2 and restoration 3 and highest in the mature forest study plot. Restoration 1 has an intermediate number of individuals and the grassland and naturally regenerating study plots are similar. Evenness (Figure 5.1c) is lowest in restoration 2 and highest in the grassland study plot. Restoration 1 and the mature forest study plot have intermediate evenness, with restoration 3 being slightly lower and the naturally regenerating study plot slightly higher. Diversity (Figure 5.1d) is lowest in restoration 1, and highest in the naturally regenerating study plot and restoration 3. Restoration 2 and the grassland plots have intermediate diversity, with the mature forest study plot slightly less.

Figure 5.1 Invertebrates. Graphs (a)-(d) show the relationship between study plot and a number of diversity indices (GL= Grassland, R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

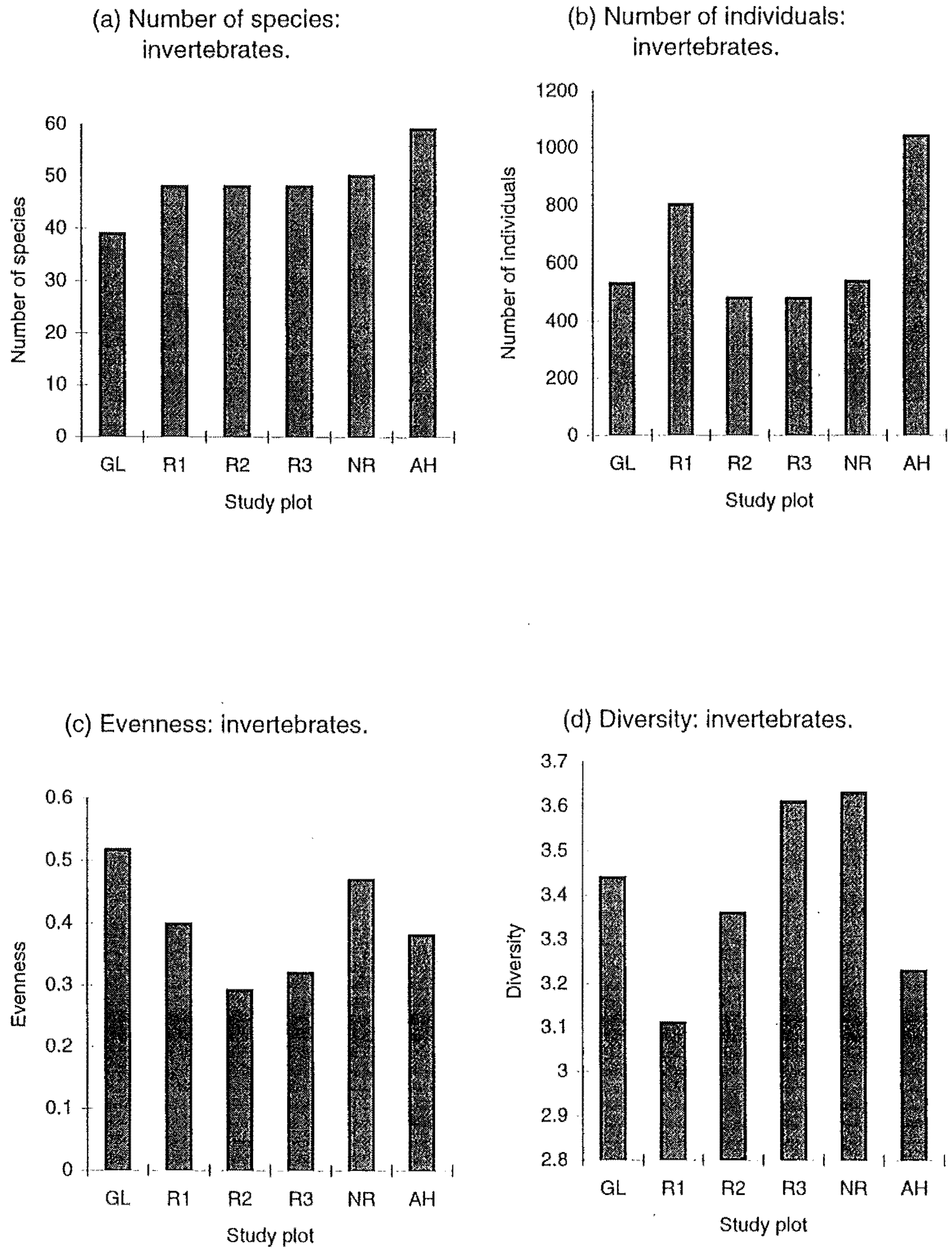


Figure 5.2 Beetles. Graphs (a)-(d) show the relationship between study plot and a number of diversity indices (GL= Grassland, R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH=Mature forest).

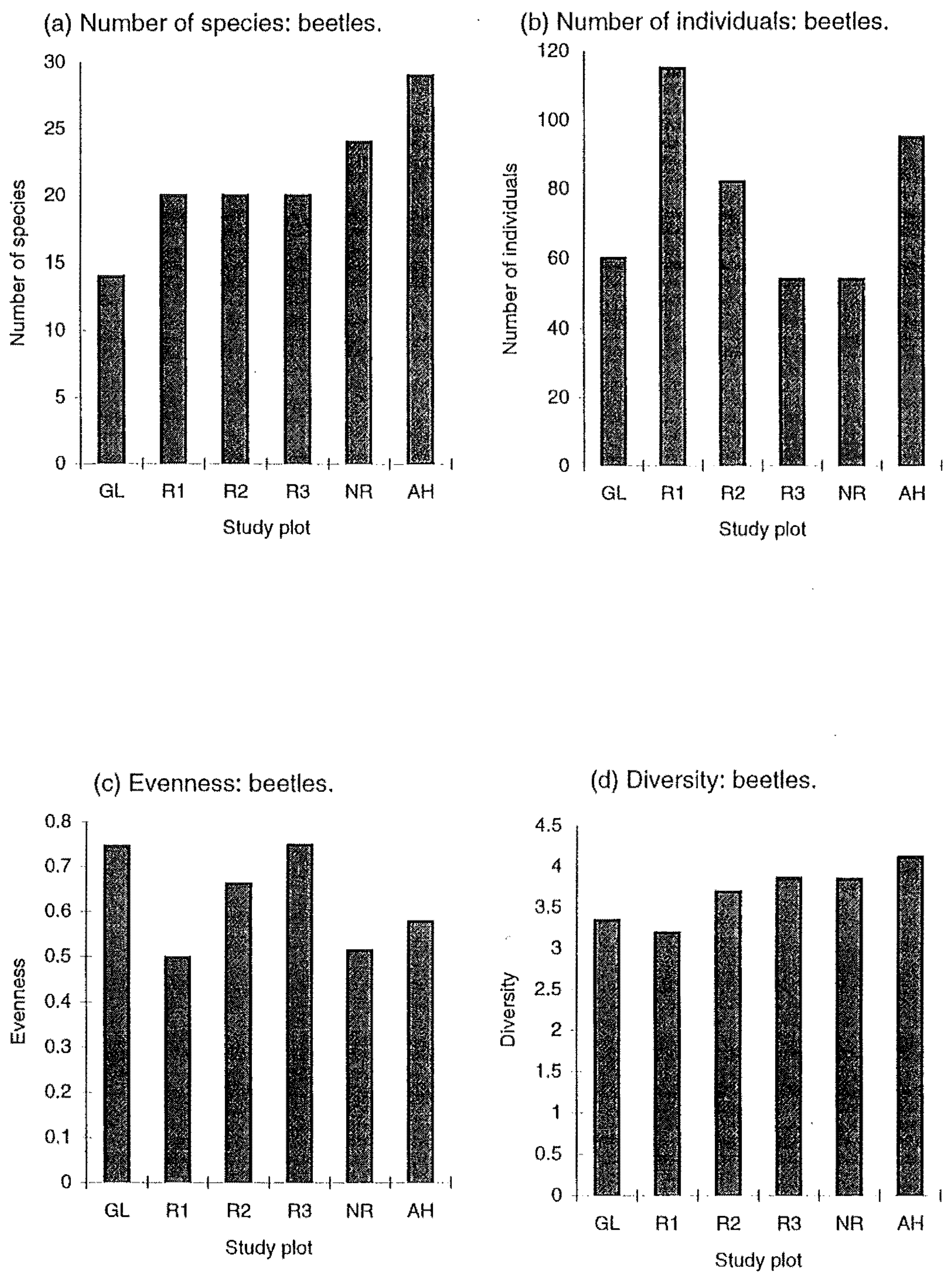
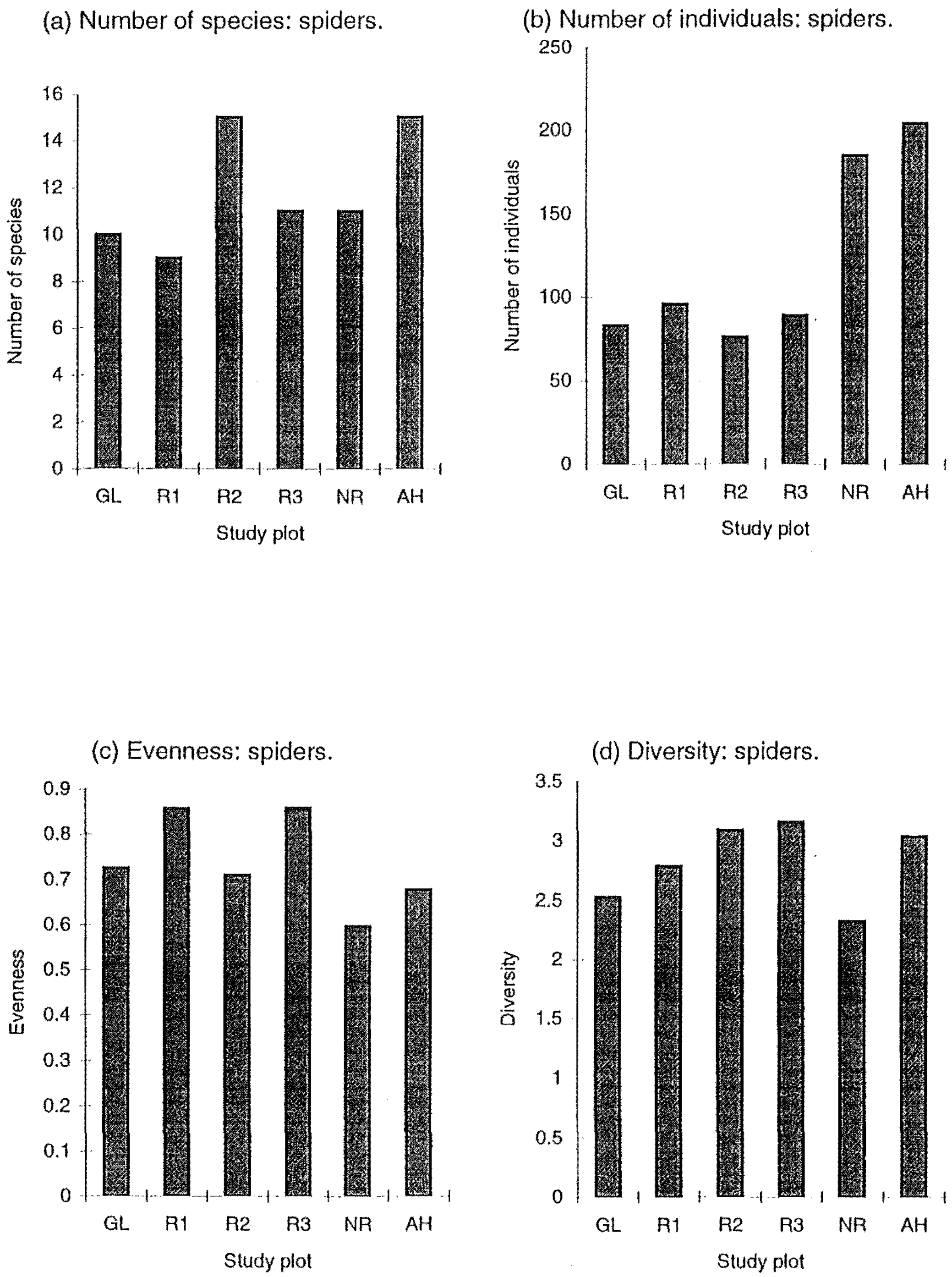


Figure 5.3 Spiders. Graphs (a)-(d) show the relationship between study plot and a number of diversity indices (GL= Grassland, R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).



5.1.2 Beetles (Figure 5.2).

Species richness (Figure 5.2a) is lowest in the grassland plot (14) and highest in the mature forest study plot (28), with the restoration plots similar. The naturally regenerating study plot has a species richness between the three restoration and mature forest study plots. The number of individuals (Figure 5.2b) is lowest in restoration 3 (52) and the naturally regenerating study plot, and highest in restoration 1 (115). Restoration 2 has an intermediate number of individuals, with the grassland and mature forest study plots slightly lower and higher, respectively. Evenness (Figure 5.2c) is lowest in restoration 1 and the naturally regenerating study plot, and highest in restoration 3 and the grassland study plot. Restoration 2 and the mature forest study plot have slightly less than intermediate evenness. Diversity (Figure 5.2d) is lowest in restoration 1 and highest in the mature forest study plot. The other two restoration study plots and the naturally regenerating study plot have intermediate diversity, with the grassland study plot slightly less.

5.1.3 Spiders (Figure 5.3).

Species richness (Figure 5.3a) is lowest in restoration 1 (9) and highest in the restoration 2 (15) and mature forest study plots. Restoration 3 and the naturally regenerating study plots have intermediate species richness, with the grassland study plot slightly less. The number of individuals (Figure 5.3b) is lowest in restoration 2 and highest for the mature forest study plot, then naturally regenerating study plots. The remaining study plots have a similar number of individuals. Evenness (Figure 5.3c) is lowest in the naturally regenerating study plot and highest in restoration 2 and restoration 3, with the remaining study plots intermediate. Diversity (Figure 5.3d) is lowest in the naturally regenerating study plot and highest in restoration 2, restoration 3 and the mature forest study plot. The remaining two study plots have intermediate diversity.

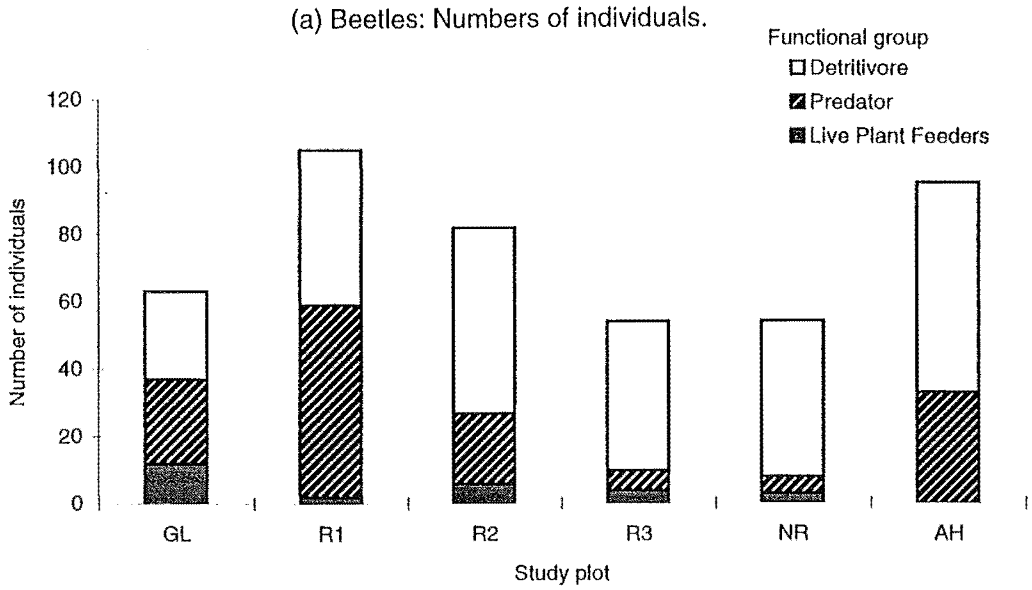
5.2 Summed abundance classes-beetles (Figure 5.4).

Detritivores make up the largest proportion of beetle individuals (Figure 5.4a) and species (Figure 5.4b) for all study plots. Predatory beetles make up the second largest proportion of beetle numbers and species, with live plant feeders featuring least. The grassland study plot has a comparatively high number of live plant feeding individuals, but these are represented by relatively few species. Conversely the naturally regenerating study plot has a comparatively high number of plant feeding species but relatively few individuals. No live plant feeding beetles were present in the mature forest study plot. The grassland and restoration 1 study plots have comparatively low numbers of predatory species containing many individuals, contrasting with restoration 3 (R3) and the naturally regenerating study plots with many predatory species but few individuals. The remaining predatory and live plant feeder, and all detritivore groups are relatively evenly represented within study plots, with either many species represented by many individuals or few species represented by few individuals.

5.3 Number of species in common (Table 5.1).

The invertebrate species in the grassland and mature forest study plots are least similar to each other (Jaccards coefficient= 0.30) while the invertebrate species in restoration 1 and the grassland study plot are most similar (Jaccards coefficient= 0.55) (Table 5.1). As the age of the study plots increases from restoration 1 to the mature forest study plot the study plot's invertebrate species similarity to the grassland study plot decreases. The invertebrate species similarity of each study plot to the mature forest study plot increases with increasing study plot age (from grassland to the naturally regenerating study plot). The three restoration study plots have more invertebrate species in common with each other than they do with either the naturally regenerating or mature forest study plots.

Figure 5.4 Summed abundance classes-beetles. Graph (a) shows the relationship between the number of individuals and functional group for each study plot. Graph (b) shows the relationship between the number of species and functional group for each study plot (GL= Grassland, R1= Restoration 1, R2= Restoration 2, R3 = Restoration 3, NR= Natural regeneration, AH= Mature forest).



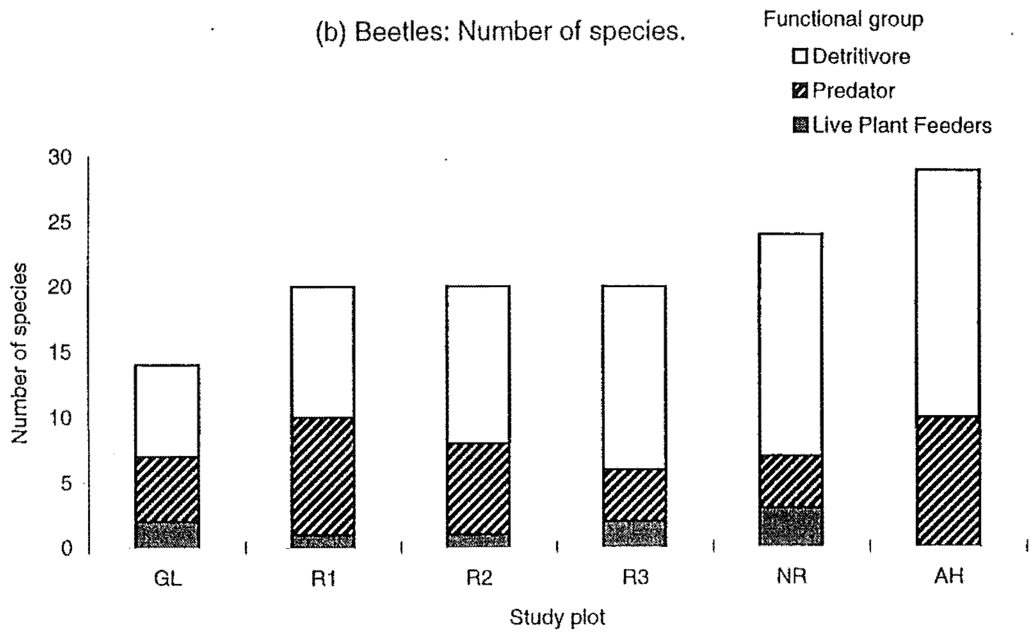


Table 5.1 Invertebrate similarity (Jaccards coefficient) between study plots.

	GL	R1	R2	R3	NR	AH
GL	1.00					
R1	0.55	1.00				
R2	0.43	0.54	1.00			
R3	0.38	0.50	0.52	1.00		
NR	0.39	0.48	0.48	0.51	1.00	
AH	0.30	0.47	0.47	0.45	0.51	1.00

GL= Grassland study plot
R1= Restoration 1 study plot
R2= Restoration 2 study plot
R3= Restoration 3 study plot
NR= Naturally regenerating study plot
AH= Mature forest study plot

5.4 Ordination.

The following are the results of DCA ordinations for the 3 invertebrate categories.

5.4.1 Invertebrates (Figure 5.5, Table 5.2).

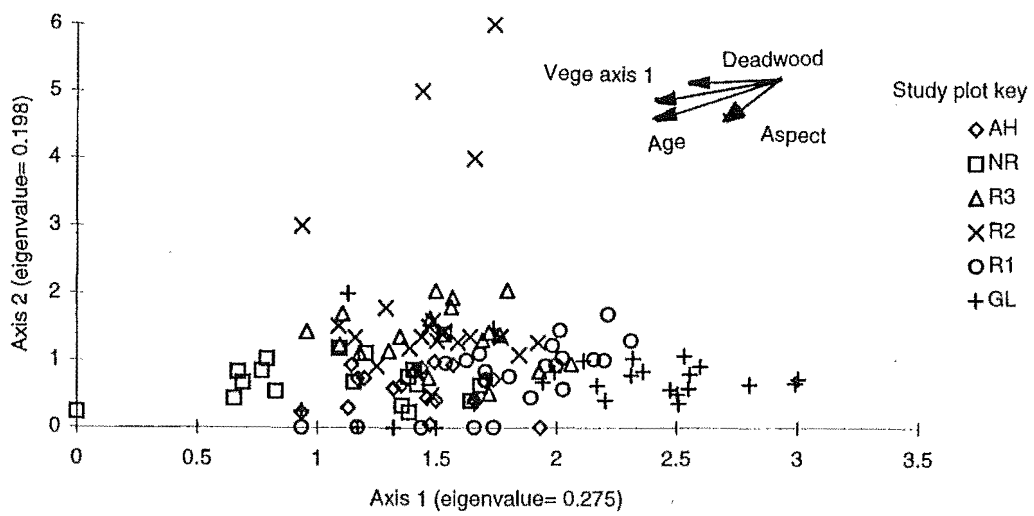
The DCA ordination (Figure 5.7a, Table 5.2a) based on total invertebrate abundance shows a slight trend with the naturally regenerating and mature forest study plots at one side of the scatter to grassland study plots on the other side, with respect to the first axis. The 1st axis (eigenvalue= 0.275) explains 6 % of the variation in the data. The 2nd axis (eigenvalue= 0.198) explains 4.4 % of the variation in the data. The gradient lengths of the 1st and 2nd axis are 3.01 and 2.05 respectively.

The second DCA ordination (Figure 5.5b, Table 5.2b) combines the invertebrate data for each study plot to try and clarify differences between study plots. This ordination suggests a clear gradient from the naturally regenerating and mature forest study plots, to the three restoration study plots, to the grassland study plot. The 1st axis (eigenvalue= 0.209) explains 35.7 % of the variation in the data. The 2nd axis (eigenvalue= 0.099) explains a further 16.9 %. The gradient lengths are 1.48 and 0.97 respectively for the 1st and 2nd axes.

Spearman rank correlation coefficients (Table 5.2c) of the invertebrate ordination (Figure 5.5a), show that the 1st invertebrate axis is strongly negatively correlated with vegetation axis 1 (-0.767) and plot age (-0.726). The 1st invertebrate axis also shows a moderate negative correlation with deadwood (-0.686). The 2nd invertebrate axis shows a weak positive correlation with slope (0.564). There are no significant correlations between the remaining environmental variables and the 1st or 2nd invertebrate axes.

Figure 5.5 DCA invertebrate ordination. Graph (a) shows DCA invertebrate scatter. Graph (b) shows DCA scatter of combined invertebrate data for each study plot. The correlation of the most important criteria are shown (length of line is proportional to correlation size) (Gl= Grassland, R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).

(a) Invertebrate ordination.



(b) Invertebrate ordination (study plot data combined).

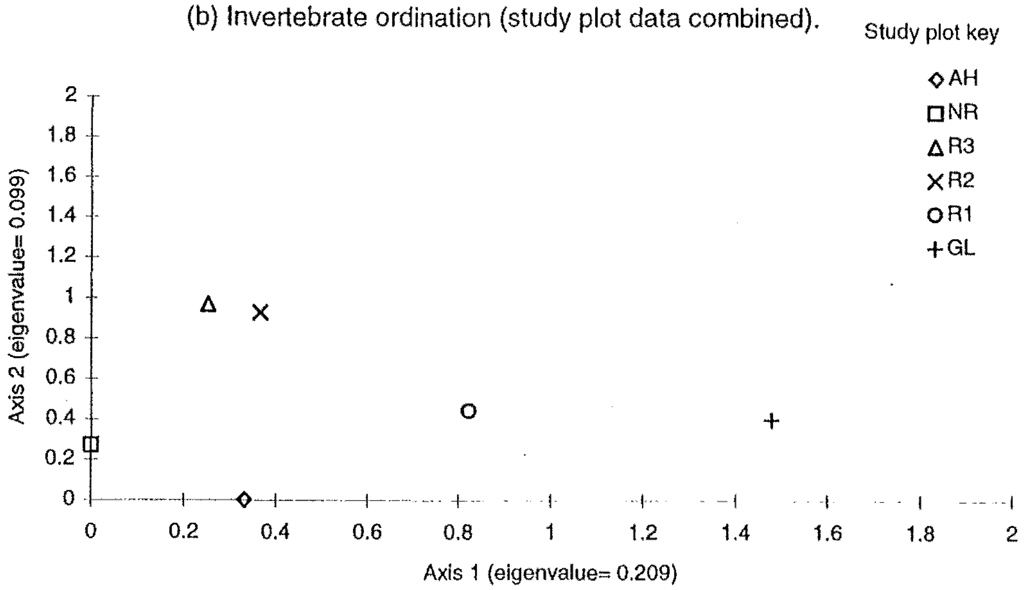
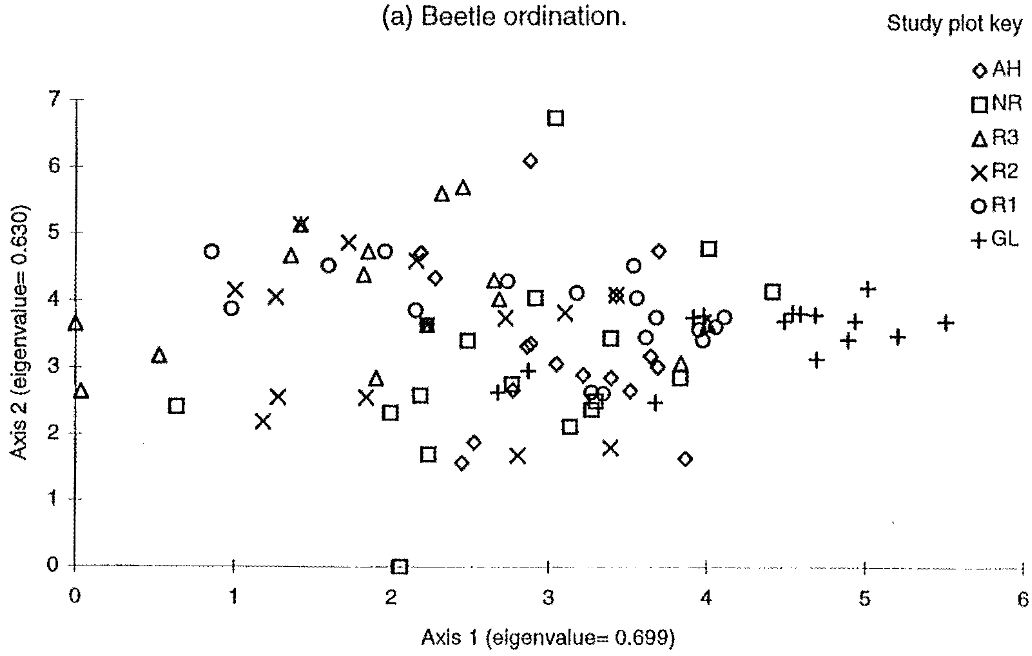


Figure 5.6 DCA beetle ordination. Graph (a) shows DCA beetle scatter with outlier Leptopinae omitted. Graph (b) shows DCA scatter of combined beetle data for each study plot (No significant correlations) (GL= Grassland, R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).



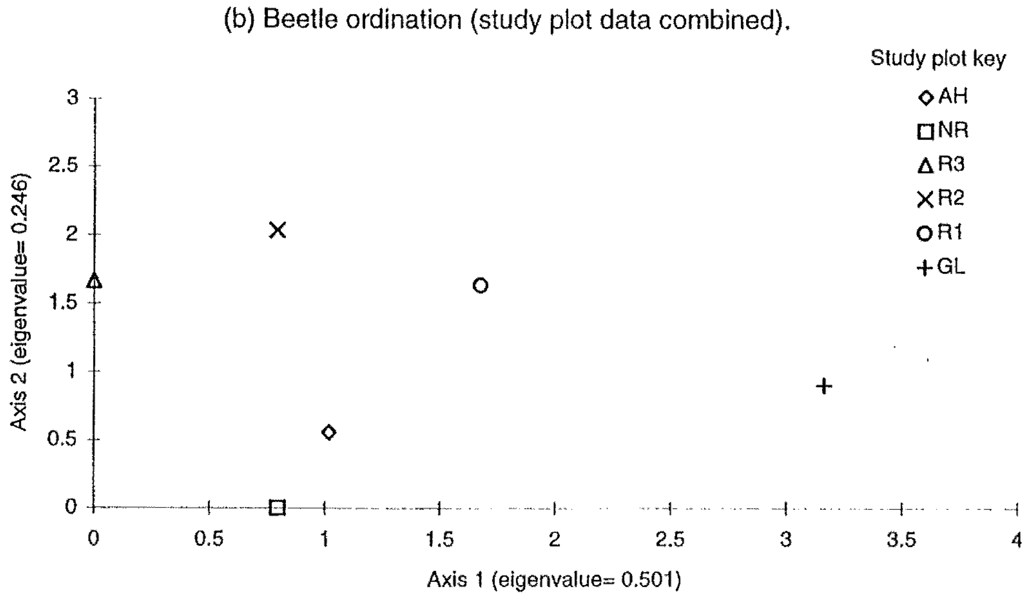
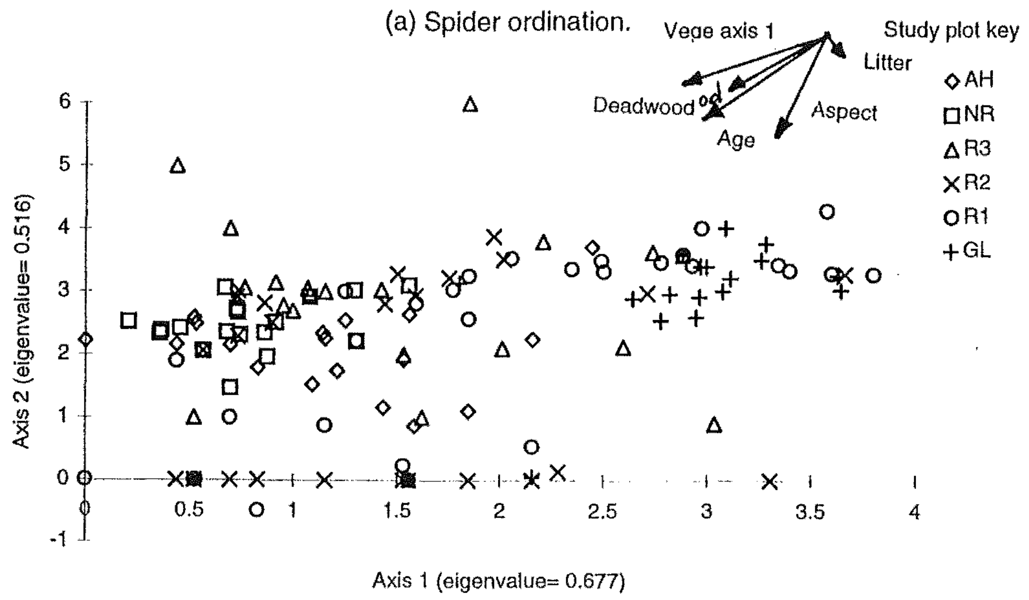


Figure 5.7 DCA spider ordination. Graph (a) shows DCA spider scatter. Graph (b) shows scatter of combined spider data for each study plot. The correlation of the most important criteria are shown (length of line is proportional to correlation size) (GL= Grassland, R1= Restoration 1, R2= Restoration 2, R3= Restoration 3, NR= Natural regeneration, AH= Mature forest).



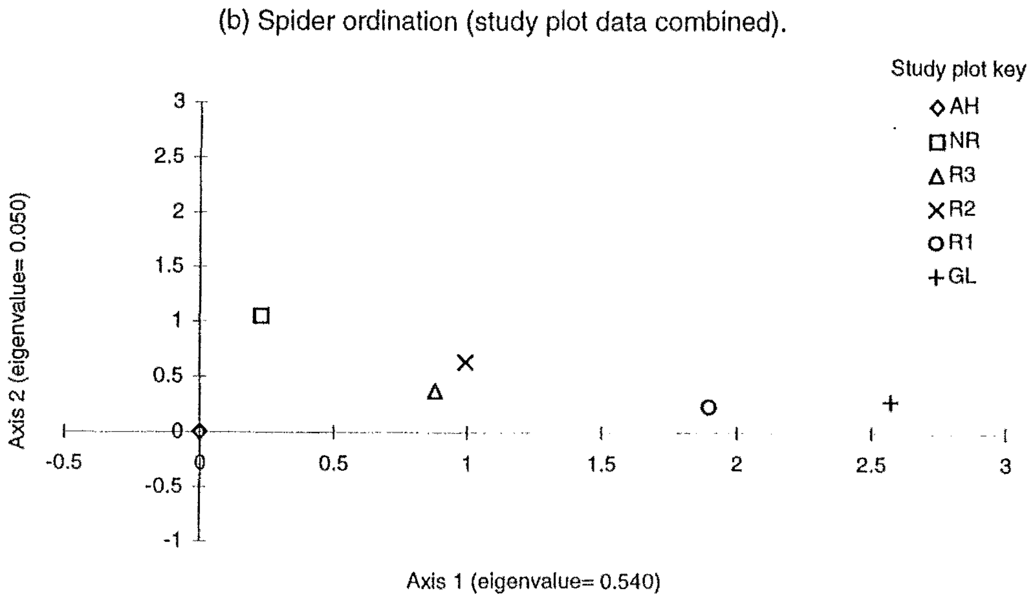


Table 5.2a Invertebrate DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.275	0.198	0.156	0.126
Lengths of gradient	3.005	2.047	2.002	2.047
Cumulative percentage variance of species data	6.0	10.4	13.8	16.5
Total sum of eigenvalues	4.563			

Table 5.2b Combined invertebrate DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.209	0.099	0.014	0.000
Lengths of gradient	1.479	0.967	0.779	0.000
Cumulative percentage variance of species data	35.7	52.6	59.4	0.00
Total sum of eigenvalues	0.585			

Table 5.2c Spearman rank correlation coefficients calculated between environmental variables and the first two invertebrate axes.

	Axis 1	Axis 2
Vegetation Axis 1	-0.767	-0.141
Vegetation Axis 2	-0.340	-0.334
Aspect	-0.466	-0.144
Slope	-0.132	0.564
Moisture	-0.410	0.185
Age	-0.726	-0.171
Litter	-0.071	0.144
Deadwood	-0.686	0.006

Table 5.3a Beetle (Leptopinae omitted) DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.698	0.630	0.537	0.453
Lengths of gradient	5.504	6.745	5.142	3.745
Cumulative percentage variance of species data	5.7	10.9	15.3	19.1
Total sum of eigenvalues	12.156			

Table 5.3b Combined beetle DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.501	0.246	0.000	0.000
Lengths of gradient	3.165	2.035	0.000	0.000
Cumulative percentage variance of species data	32.3	48.1	0.0	0.0
Total sum of eigenvalues	1.553			

Table 5.3c Spearman rank correlation coefficients calculated between environmental variables and the first two beetle axes.

	Axis 1	Axis 2
Vegetation Axis 1	-0.248	-0.206
Vegetation Axis 2	-0.049	-0.161
Aspect	-0.170	-0.150
Slope	-0.521	0.237
Moisture	-0.316	0.154
Age	-0.237	-0.190
Litter	-0.170	0.128
Deadwood	-0.302	-0.139

Table 5.4a Spider DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.677	0.516	0.362	0.264
Lengths of gradient	3.800	4.281	2.429	3.262
Cumulative percentage variance of species data	12.1	21.4	27.9	32.6
Total sum of eigenvalues	5.583			

Table 5.4b Combined spider DCA ordination summary.

Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.540	0.050	0.000	0.000
Lengths of gradient	2.572	1.050	0.000	0.000
Cumulative percentage variance of species data	52.6	57.4	0.000	0.000
Total sum of eigenvalues	1.027			

Table 5.4c Spearman rank correlation coefficients calculated between environmental variables and the first two spider axes.

	Axis 1	Axis 2
Vegetation Axis 1	-0.720	-0.516
Vegetation Axis 2	-0.382	-0.335
Aspect	-0.439	-0.527
Slope	-0.008	0.251
Moisture	-0.301	-0.316
Age	-0.678	-0.578
Litter	-0.533	0.080
Deadwood	-0.608	-0.527

5.4.2 Beetles (Figure 5.6, Table 5.3).

The DCA ordination (Figure 5.6a, Table 5.3a) based on beetle abundance (with outlier species Leptopinae omitted) fails to clearly separate the six study plots with respect to either the 1st or 2nd axes. This may be due to noise associated with invertebrate data. The 1st axis accounts for 5.7 % (eigenvalue= 0.99) of the variation in the data, while the 2nd axis accounts for 10.9 % (eigenvalue= 0.630) of the variation in the data. The gradient lengths of the 1st and 2nd axes are 5.50 and 6.75 respectively.

The DCA ordination of combined invertebrate data (Figure 5.6b, Table 5.3b) suggests a clear gradient from the naturally regenerating and mature forest study plots, to the three restoration study plots, to the grassland study plot. The eigenvalues for the 1st and 2nd axes are 0.501 and 0.246 respectively, explaining 32.3 % and 15.8 % of the variation in the data. The gradient lengths of 3.17 and 2.04 are associated with the 1st and 2nd axes respectively.

Spearman rank correlation coefficients (Table 5.3c) of the beetle ordination (Figure 5.6a), show a weakly negative correlation between the 1st beetle axis and slope (-0.521). There are no significant correlations between the remaining environmental variables and the 1st and 2nd beetle axes.

5.4.3 Spiders (Figure 5.7, Table 5.4).

The DCA spider ordination (Figure 5.7a, Table 5.4a) shows a weak separation of the grassland study plot from the other study plots, with no other obvious trends. The 1st axis eigenvalue is 0.677 (explaining 12.1 % of the variation in the data). The 2nd axis eigenvalue is 0.516 (explaining 9.6 % of the variation in the data). The respective gradient lengths of the 1st and 2nd axes are 3.80 and 4.28.

The DCA ordination of the combined spider data (Figure 5.7b, Table 5.4b) suggests a gradient from the naturally regenerating and mature forest study plots to restoration 2 and restoration 3, to restoration 1 and the grassland study plot. The first axis eigenvalue is 0.540 (explaining 52.6 % of the variation in the data). Axis 2 (eigenvalue 0.050) explains 4.8 % of variation in the data. The lengths of the 1st and 2nd axes are 2.57 and 1.05 respectively.

Spearman rank correlation coefficients (Table 5.4c) of the spider ordination (Figure 5.7a) show a strong negative correlation (-0.720) between the 1st spider axis and the 1st vegetation axis. Moderate to weak negative correlations are show between the 1st spider axis and plot age (-0.678), deadwood (-0.608), the 2nd spider axis and the 1st vegetation axis (-0.516), aspect (-0.527), plot age (-0.579) and deadwood (-0.527).

5.5 Primary interpretation of invertebrate results.

The overall picture from the invertebrate analysis is the chronological ordering of the study plots, suggesting a sequence of invertebrate community development associated with study plot development.

5.5.1 Beetles.

Both the ordination and diversity assessments of the study plots illustrate the similarities of the study plots to each other. The beetle ordination (Figure 5.6b) suggests a clear gradient from the grassland study plot, to the three restoration plots, to the naturally regenerating and mature forest study plots. The absence of significant correlations between the beetles and the measured environmental variables (Table 5.3c) gives little evidence to suggest which factors are influencing beetle community development. Similarly with the vegetation ordinations, this ordination places the three restoration study plots between the grassland study plot, and the naturally regenerating and mature forest study plots, indicating beetle community development is showing similar influences to the vegetation component of the study plots.

The diversity assessments give little information on overall beetle community developmental trends, other than an increase in species richness (Figure 5.2a) with increasing study plot age.

Summing the beetle abundance classes (Figure 5.4) gives greater interpretable information than commonly associated with a single value diversity index. While the three restoration study plots have a similar number of species (Figure 5.4b) the proportion of detritivore species increases with increasing study plot age. This trend is shown with all study plots, possibly a reflection of the greater decomposable biomass the mature forest study plot. While there is a high number of predatory species (and individuals) in restoration 1 this declines in the remaining restoration study plots before increasing in the mature forest study plot, suggesting the early stages of restoration are providing enhanced opportunities for predators.

Summary.

While the potential factors influencing beetle community development are not obvious there is a recognisable community development trend associated with study plot age.

5.5.2 Spiders.

The spider ordination (Figure 5.7b) suggests a gradient from the naturally regenerating and mature forest study plots, to restoration 2 and restoration 3, to restoration 1 and the grassland study plot, on the first two ordination axes. Spider community development is strongly correlated with the vegetation ordination 1st axis, study plot age and deadwood (Table 5.4c), again illustrating community development associated with the study plot age.

The increase in spider individuals (Figure 5.3b) in the naturally regenerating and mature forest study plots may be associated with an increase in potential prey items in these study plots, associated with more developed communities overall .

Summary.

The spider communities of the study plots appear to be following a similar developmental trend from the grassland study plot to the mature forest study plot, as observed in both the beetle and vegetation analyses.

5.5.3 Invertebrates.

The ordination of all the invertebrates in the study plots (Figure 5.5b) again illustrates the dominant pattern of community development from the grassland study plot to the mature forest study plot. The invertebrate communities in the study plots are strongly correlated with vegetation, study plot age and deadwood (Table 5.2a) providing strong evidence for the observed community development sequence.

The similarity of the invertebrates in the study plots to each other (Table 5.1) provides further evidence for the community development sequence suggested above. The invertebrate communities of the grassland and mature forest study plots are least similar. It is possible to interpret the three groupings of the study plots which appear throughout the invertebrate analysis; the grassland study plot, the three restoration study plots, and the naturally regenerating and mature forest study plots.

Summary.

While the patterns associated with community development are not as clear as those interpreted from the vegetation analyses, they are consistent with the patterns of community development observed in the vegetation analyses. Such observations strongly suggests close links between developing vegetation patterns and the development of invertebrate community patterns. These patterns suggest an invertebrate community developmental sequence from the grassland study plot to the mature forest study plot closely associated with study plot age.

CHAPTER SIX

DISCUSSION

In this chapter the findings of this study are discussed as determined by the aims suggested in the introduction and compared with other studies.

6.1 Introduction.

While conservation biology has traditionally focussed on the preservation of species, individually and collectively (Soulé & Wilcox 1980; Soulé 1986; Simberloff 1988; Wilson & Peter 1988), there has recently been a move toward conservation at higher levels of organisation (eg. ecosystems and landscapes) (Noss & Harris 1986; Noss 1983,1987,1990; Gosselink *et al* 1990; Harker *et al* 1993; Hobbs *et al* 1993; Hobbs 1994). This shift developed out of a recognition that individual species require functioning ecosystems to survive, and that nature reserves are strongly influenced by their surrounding landscapes (Diamond 1975; Janzen 1983; Lord & Norton 1990; Saunders *et al* 1991; Norton 1992a,b; Hobbs 1994; Norton *et al* 1995). Ecological restoration can provide increased habitat areas for biological conservation at this higher level of organisation.

Current ecological theory may provide the means to restore degraded lands as self sustaining natural systems, with the potential for biodiversity conservation, (Ewel 1987; Norton 1991; Main & Lambeck 1993). Although there is often confusion over what is meant by ecological restoration (Hobbs & Norton 1996) most restoration projects seem to focus on the idea of re-constructing what may have occurred (or a close approximation to this) on a site prior to its disturbance (Norton 1991; Norton 1992a; Aronson *et al* 1993a; Berger 1993; Williams 1993; Hobbs & Norton 1996). However, if it is to be successful in conserving biodiversity, restoration must go beyond the reconstruction of structure, composition and appearance of a site (Ewel 1987; Andersen 1993) and restore biological interactions, biotic processes, and integrity along with structure and composition (Majer 1989a; Andersen 1993; Aronson *et al* 1993a,b; Bradshaw 1993;

Main & Lambeck 1993; Saunders *et al* 1993; Williams 1993; Chambers *et al* 1994; Naveh 1994; Simmonds *et al* 1994; Hobbs & Norton 1996). Recreating structure and composition (eg an approach suggested by Luken (1990)) without restoring naturally occurring functions, or recreating these functions in the absence of natural structure and composition fails to constitute complete restoration (Westman 1991; Berger 1993; Cairns 1993a; Brown 1994; Hobbs & Norton 1996). This will not result in biodiversity conservation.

There are at present few established criteria for measuring the success of restoration plantings (Berger 1991; Hobbs & Norton 1996). However the need to monitor such projects is widely recognised (Norton 1992a; Andersen 1993; Bradshaw 1993; Williams 1993; McClanahan & Wolfe 1993; Robinson & Handel 1993; Saunders *et al* 1993; Chambers *et al* 1994; Simmonds *et al* 1994; Dahm *et al* 1995; Trexler 1995). A current difficulty when monitoring the success of a restoration project is that ecosystem functions and species compositions are not well understood (Armstrong 1993; Simberloff 1993; Chambers *et al* 1994; Trexler 1995; Hobbs & Norton 1996).

Aronson *et al* (1993a), Cairns (1993a) and Hobbs & Norton (1996) list key ecosystem attributes and suggest that the presence of these factors in restored sites can signify restoration success.

In the context of this study, I view restoration success as forming a continuum from the successful establishment of the initial planting through to the successful establishment of those attributes that ensure a self sustaining, functional natural system (Figure 6.1). A natural system is defined here as consisting predominantly of native species. While the later stages of this continuum are the most likely goals for restoration projects, clearly the initial stages at planting must be successful if the longer term restoration goals are to be met. For example, Majer (1989b) in identifying a number of successional pathways for animal succession, noted that the animals must first colonise the area being restored.

Figure 6.1 Restoration continuum.

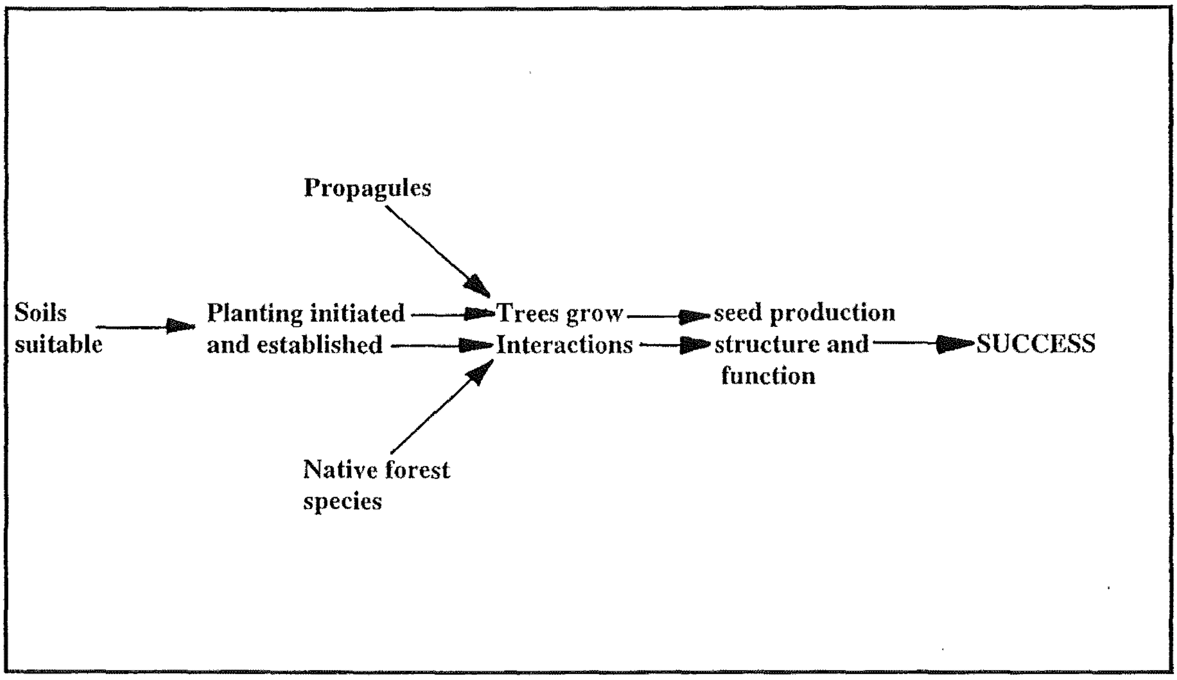
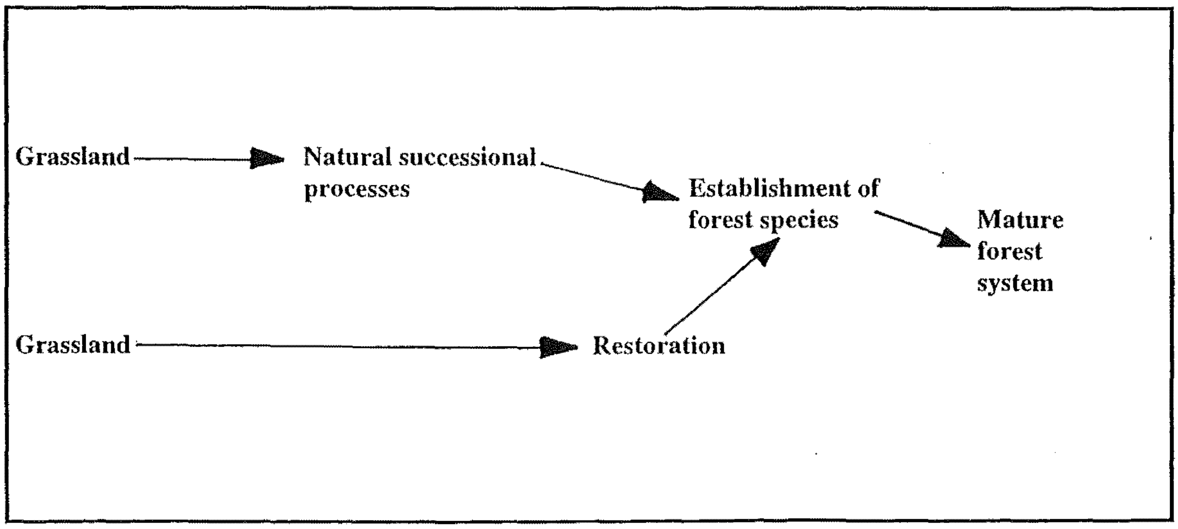


Figure 6.2 The role of restoration in natural succession.



Through success in the initial planting, restoration becomes a "tool" to conserve biodiversity by individuals recolonising into the restoration habitat. The return of any native species to restoration sites assists the conservation of native biodiversity (Robinson & Handel 1993). Following the successful establishment of those conditions required for recolonisation and establishment, individuals and species become a part of and contribute to the maintenance of the system.

By describing restoration success in the context of a continuum, evaluation of the success of a restoration planting can be achieved with greater ease. If conditions are suitable for the first stage of the restoration (recolonisation and establishment of species) then it is possible that the planting may develop to a later stage.

In order to determine how successful the Kennedy's Bush restoration project has been I have divided the discussion into two sections. The first is orientated toward monitoring the success of the initial plantings as habitats for the conservation of indigenous forest species. Those factors identified in the introduction, influencing recolonisation and establishment will be discussed. Although the first section will focus on recolonisation and establishment, it is in the context of evaluating restoration success in the later stages of the above continuum.

The second section considers the role of restoration at the later stage of the continuum. Not only concerned with the conservation of individual species, restoration is often viewed as a means to develop and conserve ecosystems. This section will consider the role of restoration planting in ecosystem conservation, recognising that this is a more holistic approach to conservation ecology. Individual species are still considered important at this level, but it is their functional role that is of significance. Addressing restoration in this way acknowledges a move from where a restoration planting is seen just as a "habitat" for maintenance of individuals, to where a restoration planting is itself an important component of the system. Those individuals within the system are therefore important for the maintenance of the system they are a part of. Ultimately this section identifies whether the three restoration plantings in this study are successful, in respect to the goal identified in chapter two. That is whether the plantings have restored fully functioning, self-sustaining natural systems (as suggested by Andersen 1993; Aronson *et al* 1993a; Simmonds *et al* 1994).

6.2 Restoration for biodiversity conservation.

This section addresses those questions outlined in the introduction relating to the ability of restoration projects to facilitate successfully early stages of restoration. In doing this I discuss some features of the three restoration plantings that have been suggested to influence colonisation and establishment of species in these early stages.

6.2.1 Are restoration plantings facilitating the re-colonisation and establishment of native forest flora and ground invertebrates?

Most restoration projects are initiated on an ad hoc, site and situation specific basis (Hobbs & Norton 1996) and are often a case of trial and error (Robinson & Handel 1993). This is likely to be the case for the Kennedy's Bush restoration plantings. It appears that those involved with designing the restoration plantings were enthusiastic volunteers. This is indicated by the choice of species used in the plantings. It is also likely that if there were any long term goals for the restoration plantings at Kennedy's Bush, they were likely to be significantly different from any goals set if the planting were to be initiated today.

As mentioned earlier, I suggest that an appropriate goal for these plantings is to restore fully functioning self-sustaining systems that are ecologically appropriate for the area, while providing enhanced opportunities for native biodiversity conservation. Within this, I suggest that some elements of previous systems should be present if they are found in intact remnant systems. It is important to acknowledge that any differences between restored and remnant systems should not limit the perceived success of the plantings, provided they satisfy those conditions of the above goal.

The establishment of natural ecosystems by ecological restoration requires that the initial stages of the restoration be successful. Beyond the survival of those plants used in the initial planting, the plantings themselves must facilitate recolonisation and establishment of native species. Robinson and Handel (1993) suggest that a likely explanation for the absence of natural succession on degraded sites is that the appropriate seeds never arrive. Microsite limitations impose "filters" on developing communities. Such filters may include competition, predation and herbivory of seeds (Robinson & Handel 1993). Majer (1989b), by recognising that colonisation of fauna is required before any faunal succession occurs implies that a restoration planting must be able to facilitate this recolonisation.

Historically, the focus on the determinants of plant community structure has largely been on post germination processes (McClanahan & Wolfe 1993). Recently it has been recognised that colonisation is an important factor contributing to community structure and consequently may limit restoration success (McClanahan & Wolfe 1993). The rate at which species colonise depends on the nature of the area to be restored, the characteristics of the species themselves and the proximity of propagule sources (MacArthur & Wilson 1967, Bradshaw 1983; Majer 1989a,b, 1990).

Both forest invertebrate and plant species are present in all three restoration study plots at Kennedy's Bush. This implies that native biodiversity is returning to the restoration study plots. The presence of regeneration tree species in all three restoration study plots indicates that the plantings are successfully re-establishing forest vegetation. This is highlighted by species regenerating which are not present in the canopy and subcanopy of the restoration study plots (Figure 4.3a). Despite the low numbers of canopy and subcanopy species in the restoration study plots (Figure 4.2a), the number of regenerating species is high indicating an increase in biodiversity.

The age of the restoration plantings appears to be an important factor influencing the amount of regeneration cover. The two older restoration study plots have the greatest regeneration cover and species richness (Figure 4.3a,b). This indicates that these plantings provide better habitats for the regeneration of potential canopy tree species and/or that long time periods are required for the establishment of new species. In addition to abundant recolonisation of tree species the two older restoration plantings also have a prominent ground cover dominated by forest species. This includes ferns, herbs and grasses (pers. obs.). These species are absent or only poorly represented in the grassland and youngest restoration study plot.

The establishment of plant communities in restoration is generally given the greatest attention (Viert 1989; Majer 1990; Samways 1993; Simmonds *et al* 1994; Hobbs & Norton 1996) with animals at best receiving secondary attention (Majer 1990). However, performance criteria involving the measurement of plant community dynamics alone is an insufficient indicator of ecosystem maintenance or stability (Williams 1993). It has often been assumed that reintroducing vegetation into an area will result in the return of the remaining ecosystem components (Hobbs & Norton 1996), especially fauna (Viert 1989; Williams 1993). While plant cover is of importance to developing fauna, the provision of adequate floristic and structural diversity and the presence of logs and litter are important for the full range of native animal recolonisation (Majer 1989a,b).

The focus on vegetation in restoration has meant that the importance of fauna in ecosystems is often understated (Berger 1991; Simmonds *et al* 1994). The establishment of fauna in restoration plantings is vital if ecosystem functions are to be reestablished (Majer 1989a, 1990; Andersen 1993; Robinson & Handel 1993; Williams 1993; Simmonds *et al* 1994; Hobbs & Norton 1996). Invertebrates represent a huge component of biodiversity (Southwood 1978; Price 1984; Wilson 1987; Wilson 1988; Gaston 1991; Samways 1993) and play substantial roles in the functional aspects of ecosystems (Southwood 1978; Price 1984; Wilson 1987; Majer 1989a; Andersen 1993; Kim 1993; Samways 1993; Williams 1993), contributing to such processes as pollination, seed dispersal, decomposition, predation, herbivory and nutrient cycling to name a few. In fact, the most common biotic interaction on Earth is between insects and plants (Samways 1993).

Due to their valuable contribution to ecosystems, fauna, especially invertebrates, provide important indicators to monitor restoration success (Refseth 1980; Louda 1988; Majer

1989a,b; Hutcheson 1990; Andersen 1993; Kremen *et al* 1993; Williams 1993; Simmonds *et al* 1994). In addition to their contribution in the functioning of ecosystems, invertebrates with short generation times and large reproductive outputs (Williams 1993) provide a more sensitive indication than plants of the overall state and well being of an ecosystem (Andersen 1993). Also, the vast diversity (Wilson 1987; Samways 1993) of invertebrates provides a potentially wide array of probes for use in monitoring virtually any ecological situation (Kremen *et al* 1993).

Despite the significant grassland invertebrate species component present in the restoration study plots (Table 5.1) the forest invertebrate component is greater. While the number of forest invertebrate species is similar in the three restoration study plots, the reduction in the number of grassland species as the restoration study plots age indicates that the restorations become less suitable for grassland invertebrate species as they mature.

The absence of a greater invertebrate component as the restoration study plots age, and also the mature forest and naturally regenerating study plots, is likely to be due to a flaw in the trapping technique. As the study plots develop greater structural diversity the invertebrate component will move to take advantage of these changes. Here it is likely that much of the invertebrate fauna of the more developed restoration, the naturally regenerating and mature forest study plots are living above the ground where a greater diversity of resources is found. Therefore the invertebrates captured by pitfall trapping are not necessarily representative of all the species present.

All three restoration plantings studied at Kennedy's Bush appear to be successfully facilitating the re-colonisation and establishment of native biodiversity in the form of forest vegetation and ground invertebrates. Despite the very low regeneration cover in restoration 1, the presence of plant species (other than those found in the canopy) and invertebrate species found only in forested areas, suggests that initial recolonisation of some species into restored sites may occur rapidly in this area. The extent of recolonisation of invertebrate and plant species in the two older restoration study plots indicates that initial developmental stages of these restoration study plots were successful. Furthermore, the plantings in these study plots are successfully initiating the recolonisation and establishment of further native forest biodiversity.

6.2.2 Does it matter what species we plant to initiate the recolonisation and establishment of native forest species?

The current paradigm in restoration ecology involves returning a degraded system to a desired state by accelerating or reinstating successional processes (Ashby 1987; Bradshaw 1987, 1989; Uhl 1988; Majer 1989a; Luken 1990; Hobbs & Norton 1996). In all but the most degraded sites, successional processes should return a system to near its natural state with a minimum of human interference (Norton 1991; Aronson *et al* 1993a,b). It is usually assumed that correct species selection in the initial planting is essential to facilitate and condense the early stages of natural successional processes (Aber 1987; Norton 1991, 1992; McClanahan & Wolfe 1993). Identification of such species is best done by studying natural successional patterns in unmodified vegetation of sites with vegetation similar to the intended end point of the restoration (Jordan *et al* 1987; Norton 1991,1992; Williams 1993). Hobbs & Norton (1996) highlight confusion in the literature regarding the use of reference ecosystems for restoration. They consider using reference ecosystems essential to guide the restoration (ie what species may be appropriately used for planting), but that the use of reference systems in this way does not imply a goal for the outcome restoration (Hobbs & Norton 1996).

Norton (1991) suggests that while species choice is important, the species planted initially are unlikely to dominate the eventual vegetation cover. Instead they provide conditions that facilitate establishment of other species, with the final outcome of the restoration being largely dictated by nature. Others (eg. MacMahon 1987; Chambers *et al* 1994) suggest that edaphic conditions and the species established at the time of reclamation may have lasting effects on future ecosystem trajectory. McClanahan and Wolfe (1993) suggest that although planting desired species may accelerate reforestation of late-succession, high diversity ecosystems, the resulting system may be different (in terms of diversity and processes) from what may be organised by nature. They do suggest however, that it may at times be beneficial to enhance the attractiveness of sites to animals that disperse propagules of late-successional species (Nepstad *et al* 1991; McClanahan & Wolfe 1993). In cases where sites are extremely disturbed (eg. soil must be recreated before plant community can function) certain plant species (eg. nitrogen fixing) may be used to ameliorate conditions so normal successional processes can become established (Aber 1987; Armstrong 1993; Saunders *et al* 1993).

If initial species choice does not play a significant role in determining which species colonise and establish into restoration plantings, this may strongly implicate the use of non-native species, especially if some feature of non-native species makes them more attractive than natives (eg. cheap, easy propagation, fast growing).

One of the principle threats to the successful conservation of natural ecosystems is the invasion of non-native species and their impacts on native biota (Norton 1992a,b; Drake *et* al 1993; Loope & Medeiros 1994). This problem has been considered in the context of restoration by many authors (eg. Ashby 1987; Aber 1987; Ewel 1987; Luken 1990; Norton 1991; Berger 1993; Harker *et al.* 1993; Hobbs & Mooney 1993). Invading species may act as modifying influences on successional trajectories of restoration plantings and may interfere dramatically with intended ecosystem processes (Hobbs & Mooney 1993). Invaders are also known to disrupt ecosystem properties through competition, predation, introduction of disease vectors or by affecting mutualistic relationships (Berger 1993) and can lead to the extinctions of local native species (Luken 1990).

Non native species can pose a major threat for ecological restoration because they are often aggressive and can overwhelm native species, thus altering ecosystem structure (Berger 1993). Ecosystems that are most susceptible to invasion are those with breaks in the natural plant cover. The open nature of early restoration areas means colonisation and spread of invaders may be faster than in closed vegetation (Berger 1993; Hobbs & Mooney 1993).

In some cases, introduced species may play an important role in restoration (Aber 1987; Norton 1991; Hobbs & Mooney 1993). The extent to which this occurs is dependent on each particular project. Where the aim of a project is to restore a system to a state as near to natural as possible, the use of introduced species should be considered generally unacceptable (Hobbs & Mooney 1993). Those cases where non native species may be valuable are in heavily altered systems where native plants are unable to establish, or where appropriate native species have become locally extinct (Hobbs & Mooney 1993). Such species may also be used to stabilise sites, as nurse crops for slow growing native species, to restore impaired ecosystem functions, or to sequester nutrients from sites with elevated nutrient levels (Aber 1987; Hobbs & Mooney 1993; Recher 1993).

In a highly modified environment such as the Canterbury Plains, influence from exotic species is unavoidable. Initial restoration processes will be hampered by invasive weeds, especially pasture grasses, which inhibit native plant regeneration. Once a forest canopy has formed, grasses and other introduced weeds, such as gorse and broom are unlikely to threaten to the restored system.

The restoration study plots for this study were planted with apparently little consideration of the use of local genetic material as suggested by Norton (1991) and Saunders *et al* (1993). The dominant component of the planting, *Olearia paniculata* is not a species that plays a dominant role in natural successional process of the area and the plants used in the planting are of North Island provenance (Kelly 1972; Simpson 1992; Wilson 1992).

The canopy and subcanopy vegetation layers of the restoration study plots (Figure 4.6) are floristically dissimilar from the natural regenerating and mature forest study plots. Similarly, the canopy and subcanopy vegetation layers differ between the three restoration study plots (Figure 4.6). As the canopy and subcanopy layers of the restoration study plots are likely to consist largely of planted species (D. Norton pers. comm.) the separation of these study plots on the ordination scatter (Figure 4.6) indicates that the planting regimes of the study plots differ. While the dominant planted species is *Olearia paniculata* in all three of the restoration study plots, other species were also planted and their abundances differ between study plots. The reasons for the planting of the other species are not known. It is possible that some of the species thought to be planted have actually been naturally established. However it is likely that such establishment is very localised and would be an insignificant component of these canopies.

Despite the floristic differences in the canopies, and in contradiction to the above concerns regarding initial species selection determining the restoration trajectory, the differences in the three restoration study plots canopy floristics do not appear to correspond to differences in species recolonisation and establishment (based on the regenerating tree species within each restoration study plot) (MacMahon 1987; McClanahan & Wolfe 1993; Chambers *et al* 1994). That the regenerating canopy vegetation of the restoration study plots is very similar floristically (Figure 4.7a,b) to the regenerating canopy vegetation of the naturally regenerating and forested study plots suggests that the species used in the initial restoration plantings are facilitating the recolonisation and establishment of similar regenerating species that would be expected to occur during natural succession. From this I suggest that those species chosen for restoration in these sites are unlikely to hinder restoration progress. This is discussed further in the next section.

The absence of tree seedlings from grassland areas suggests that regeneration into grass is unlikely. Aronson *et al* (1993a) and Hobbs & Norton (1996) use the concept of thresholds to explain this phenomenon. When a system is in a degraded state and crosses a threshold, removal of the degrading influence will not be sufficient to allow a transition back to something approximating the original state (Wilson & Agnew 1992; Aronson *et al* 1993b; Hobbs & Norton 1996). Hobbs & Norton (1996) suggest that such transitions in states may be more difficult to force when the transitions involve changes in ecosystem composition in terms of the functional groups present. A move from one vegetated state to another, ie grassland to shrub land, can be a very difficult transition to force, compared to that between, for example, different grassland types (Hobbs & Norton 1996).

In the Port Hills environment, the transition from grassland to shrubland is likely to be a difficult transition to force due to two main reasons: (1) Any seed dispersed by forest tree species into grassland is unlikely to reach suitable germination sites (Allen *et al* 1992) and (2) if germination were possible, most tree species are unable to grow in such a competitive environment (Allen *et al* 1992), especially given the dominant influence of desiccation. The Port Hills in summer is dry (McGann 1983; Jayet 1986; Innes & Kelly 1991). Dry winds, after blowing across the plains, are likely to be the greatest desiccation factor for young seedlings (J. Hutcheson pers. comm.).

The absence of any regenerating tree species from grasslands indicates that canopy cover is required for the regeneration of native tree species (Allen & Hoekstra 1987; Allen *et al* 1992; Wilson 1994). Canopy cover removes the competitive pasture grasses and provides shelter from desiccation while providing other conditions that appear to be required for the establishment of forest species (eg litter, moisture and appropriate light).

There has been some criticism of the use of *Olearia paniculata* in these plantings (Simpson 1992), however the re-colonisation of both vegetation and invertebrates shows that it is an effective nurse species. It appears to be a fast growing, hardy species, which quickly forms a single tree canopy that initiates regeneration. The absence of regenerating *Olearia paniculata* suggests that under the current conditions it is unlikely to become a significant component of the restored forest ecosystem (Partridge 1989).

Few restoration projects are designed to consider the role of organisms other than plants (Majer 1990; Samways 1993; Williams 1993; Simmonds *et al* 1994). Others expect that once vegetation establishes the new habitat will be subsequently colonised by other organisms (Jordan *et al* 1987; Norton 1991; Williams 1993; Chambers *et al* 1994). Viert (1989) suggests where fauna have recolonised a disturbed area it is usually incidental and restoration design should encourage the return of native fauna. Although plant cover is important to developing fauna, the provision of adequate structural and floristic diversity and logs and litter is significant (Southwood *et al* 1979; Lawton 1983; Majer 1989a,b; Simmonds *et al* 1994). Saunders *et al* (1993) and Lefroy *et al* (1993) suggest studies of the use of fauna in restored areas are needed as few data suggest revegetation aids the conservation of native animals.

While there is evidence of recolonisation and the establishment of invertebrates in the three restoration study plots at Kennedy's Bush, the composition of the recolonised invertebrate communities does not appear to be influenced in a similar manner as the recolonising vegetation component of the restored study plots (Figure 5.5). This does not imply that differences in the species used in the initial planting are influencing invertebrate community patterns, but that invertebrates are sensitive to factors other than the planted vegetation. While the invertebrate communities are correlated with vegetation (Table 5.2c), vegetation appears to be largely a function of age (Table 4.4b). The strong relationships between invertebrates and age, deadwood and litter (Table 5.2c) indicates that invertebrates are responding to the overall development or maturity of the study plots. The trends across the axes of the invertebrate ordinations (Figures 5.5 to 5.7) imply that invertebrates are showing successional changes in community structure. Therefore to identify accurately if the species used in the initial restoration planting are influencing invertebrate community composition, experimentation with different species at the time of planting may be required. However, as the invertebrate component of the restored study plots shows a strong correlation with the plant community, and since the vegetation of the restored study plots appears to be developing in a similar fashion, these differences in invertebrate communities may not be a response to differences in initial invertebrate colonisation and establishment, but a function of ecosystem change since planting.

In the restoration plots in this study there is a prominent component of non native species. Luken (1990) suggests there are in fact few nature reserves immune from the influence of introduced species. While most restoration designs are influenced by the native vegetation of the area (Jordan *et al* 1987; Williams 1993; Hobbs & Norton 1996) this does not appear to have happened at Kennedy's Bush. As the dominant component (*Olearia paniculata*) of the restoration plantings is not commonly associated with natural succession in the area, this study provides a unique opportunity to test whether those species used in initial restoration plantings greatly influence the colonisation and establishment of native forest species, and whether it is necessary to plant species associated with the early stages of natural succession. *Olearia paniculata* appears to have been successful as a nurse crop and in many ways is an ideal species to be used for restoration in the Port Hills environment. It is a hardy, fast growing and drought tolerant species which successfully encourages re-colonisation of forest species.

The results indicate that any tree species with characteristics similar to *Olearia paniculata* is likely to be suitable for restoration in this area. The analysis of both the canopy and regenerating vegetation's suggests species choice alone is unlikely to play a major role in determining which tree species regenerate. While there is doubt as to how differences in planting regime influence invertebrate communities, all three planting regimes are facilitating the recolonisation and establishment of native forest plant and invertebrate species. It appears from the absence of similar forest species establishment in grasslands that canopy cover is required to induce a change in vegetated state (as suggested by Hobbs & Norton 1996), with the colonisation of components of forest communities as a consequence. It is not possible to tell whether *Olearia paniculata* is any better or initiates faster forest species recolonisation than any other species or combination of species that may have been used in these restoration plantings as there are no other planting regimes available for comparison.

6.2.3 <u>Is it necessary to plant fruiting tree species to attract birds?</u>

Despite suggestions by Majer (1989a) and others that restoration design often fails to take into account colonisation and establishment of organisms other than plants, it has also been suggested that enhancing the attractiveness of restoration plantings to birds may increase the dispersal of propagules into a site (McClanahan & Wolfe 1993; Robinson & Handel 1993). Robinson & Handel (1993) showed seed dispersal was a limiting factor in the regeneration of degraded lands in urban areas. By introducing trees and shrubs to the area with the intention of attracting birds, regeneration was enhanced to the extent that 95 % of the regenerating species were from external seed sources. Of these 71 % were fleshy fruited, bird dispersed species (Robinson & Handel 1993).

Studies have shown that seed input to reestablishing sites increased with the age of sites due to an increase in bird perch sites (Luken 1990). McClanahan & Wolfe (1993) found that seed deposition abundance and the diversity of bird dispersed plants increased under bird perches and suggested that such structures have a limited ability to enhance plant diversity under conditions of natural succession. Debussche *et al* (1982) showed that frugivorous birds were attracted from edges to the interiors of abandoned orchards by senescent fruit trees. This resulted in characteristic vegetation colonisation patterns in relation to perch trees.

The importance of frugivorous birds for seed dispersal in New Zealand was recognised from the time of first European settlement (Burrows 1994a), but the emphasis on the degree of interdependence of the plants and the animals which use them is more recent (Clout & Hay 1989; Lee *et al* 1991). Overseas studies show that wherever fleshy fruits are abundant in forests, forest regeneration processes are closely linked to frugivorous birds and other vertebrate seed dispersers (Burrows 1994c). Many New Zealand bird species appear to have co-evolved with forest tree species (Burrows 1994a). Approximately two-thirds of the native woody plant species of Banks Peninsula have fruit or seeds with some kind of fleshy attachment, most commonly either a drupe or a berry (Burrows 1994a). The abundance of fleshy fruitiness in New Zealand seeds has arisen due to two factors. Firstly, many of our plants are closely related to tropical or subtropical fleshy fruited species and their phylogeny predisposes them to fleshy fruitiness (Burrows 1994a). Secondly, the large number of small efficient fruit eating, seed dispersing birds has reinforced the fleshy fruited habit (Burrows 1994a). The flesh covering of a seed preserves it and often inhibits it from germinating until it is removed. Antibiotic properties of the fleshy tissues prevent fungal and bacterial growth on the pericarp and seed. These properties extend the shelf life of the seed, assisting temporal dispersal and keeping the fruit in an edible condition longer (Burrows 1994a). The small size of the fruit appears to be related to the gape size of many frugivorous birds (Burrows 1994c). The apparent mutualism between many forest plants and birds and the high level of importance of frugivorous birds for seed dispersal in New Zealand may be why Norton (1991) suggested that planting fruiting species in restoration projects may enhance site attractiveness and hence colonisation and establishment.

The majority of regenerating tree species in the restoration and forested study plots are bird dispersed (Figure 4.4a, Table 4.1). Regeneration in the mature forest and naturally regenerating study plots consists almost entirely of bird dispersed species. The trend of increasing proportions of bird dispersed species in the regeneration layer of restoration study plots (Figure 4.4b) suggests that birds prefer older, more established sites to reside in and hence disperse seed. The relative lack of bird dispersed regeneration in restoration 1 supports this. It is clear from the results (Figure 4.4a,b) that birds play an important role in the re-colonisation of plant species. The level of bird dispersed species in the regenerating vegetation layers of the restoration study plots and the similarity in regenerating species between the study plots, suggests that avian dispersers are playing a vital role in the regeneration processes of these study plots.

The dominant tree in the canopy and subcanopy vegetation of the restoration study plots, *Olearia paniculata*, does not produce edible fruit attractive to frugivorous birds. It appears then that avian dispersers are attracted by other features in the plantings. Perches created by planted trees are an obvious attraction (Robinson & Handel 1993; McDonald & Stiles 1983; McClanahan & Wolfe 1993). Robinson & Handel (1993) found perches positively influence the recruitment of regenerating species and that larger trees had a greater proportion of recruits beneath them than shrubs. Restoration 1 has a much reduced avian dispersed regeneration component in comparison with the other study plots. The individual trees in this study plot are still in the shrubby stage and do not appear to make ideal perch sites. Although there is a greater proportion of wind dispersed species in the regeneration layer of restoration 1 this is not due to a large amount of wind dispersed regeneration. The total regeneration cover of restoration 1 is very low (Figure 4.3b), and the wind dispersed component is no greater than in the other study plots.

Birds which disperse seeds are highly dependent on the forest in various ways and seldom travel far from it. Any seed ingested in the forest is mainly dispersed within the forest (Burrows 1994b). Kereru (*Hemiphaga novaeseelandiae*), however are an exception and undertake flights of at least 2 km between forest patches. Seeds may be excreted in open areas during such journeys, however kereru habitually rest for long periods of time on favourite perches, beneath which large seed filled faecal masses accumulate (Burrows 1994a). While some birds are dependent on fruit (Burrows 1994a) which is abundant in forest fragments like Ahuriri Scenic Reserve, there is a significant bird presence in the two older restoration study plots, the canopies of which have relatively little fruit. These restoration study plots may be providing other forms of sustenance, for example, nectar and invertebrates which benefit frugivorous birds (Burrows 1994a). While native bird numbers have significantly declined since European colonisation so has the forest. The restoration plantings may be providing much required habitat.

The forest destruction and decimation of the bird fauna of the Banks Peninsula forests have severely disrupted the native bird-forest symbiosis. It has been estimated that 26 forest bird species had disappeared from Banks Peninsula from before 1800 (Burrows 1994a), consequently the present day seed-dispersing avifauna on Banks Peninsula are few in numbers of species and individuals (Clout & Hay 1989; Burrows 1994c).

It is clear that avian dispersers are vital for forest regeneration processes (Burrows 1994a) and play an important role in the re-colonisation of plant species into the restoration study plots in this study. As the dominant canopy portion of the restoration study plots lacks any fruit to attract birds, it appears that perch sites may be an important factor in attracting these dispersers. Other features of the restored study plots that potentially attract dispersers are the presence of invertebrates, nest sites and nectar. Associated with this, the close proximity of seed and fruit at Kennedy's Bush means that bird dispersed regeneration is significant. Regeneration success in this study was dependent on the local fruit source. The outcome of the restorations may have been quite different in the absence of close native forest remnants. The proximity of remnant vegetation to potential restoration areas may be an important area for future research. It appears that fruiting tree species are not necessary to attract frugivorous birds to the restoration study plots, as the regeneration of bird dispersed tree species is abundant. Enhancing the attractiveness of a restoration study plot to frugivorous birds may be beneficial but this can not be quantified from this study.

The high component of bird dispersed species in the study plots indicates, firstly the importance of frugivorous birds as dispersers in these systems, and secondly, despite a significant reduction in the native avifaunal component of these systems regeneration is abundant. This indicates that introduced bird species may be playing a significant role in the dispersal of plant species in these systems. Introduced species such as black birds (*Turdus merula*), silvereyes (*Zosterops lateralis*), song thrushes (*Turdus philomelos*) and starlings (*Sturnus vulgaris*) may now play a compensatory role of the lost seed-dispersing birds with a small gape (Clout & Hay 1989; Burrows 1994a).

Without blackbirds and silvereyes, the link between the birds and seed dispersal in these forests would now be insubstantial (Burrows 1994c). The regeneration and re-colonisation of plant species by birds into the restoration study plots indicates that the introduced avifauna is contributing to restoration success alongside their native counterparts. These introduced bird species may also be assisting with important ecological processes such as pollination and predation (of invertebrates) (Burrows 1994a).

6.2.4 Summary.

The data analysis shows that all three restoration plantings are initiating the recolonisation and establishment of native forest plants and ground invertebrates. Species choice does not appear to be essential when considering the success of the planting. Some species or species combinations may be better or more appropriate than others but this was not considered in the context of this study. It appears that any plant species that can satisfy conditions suitable for the recruitment of forest species may be used in the Port Hills environment. In more xeric or isolated environments initial species choice is likely to be an important consideration. Birds are of great importance as dispersers of forest tree species. It appears that tree species used in restoration plantings do not need to have edible fruit in order to attract birds. Birds appear to be satisfied by perch sites and other features provided by the restoration plantings and will visit restoration study plots as a result. In such a highly modified landscape as found in the

Port Hills, influence by non-native species is unavoidable. Certain non-native species (eg. birds) are considered to be important in the restoration of this area.

All three restoration study plots are successfully facilitating the colonisation and establishment of forest species. Thus, at the very minimum these plantings are providing valuable habitats for the conservation of native biodiversity. However it is likely that the two oldest restoration study plots at least, are moving toward the later stages of the restoration continuum proposed earlier in the discussion. This will be discussed further in the next section.

6.3 Restoration for ecosystem conservation.

This section discusses whether the restoration plantings have restored natural systems required by the goal for restoration proposed earlier, that is, restoration should restore fully functioning self-sustaining systems that are ecologically appropriate for the area, while enhancing opportunities for native biodiversity conservation. While the previous section discussed the role of the restoration plantings in colonisation and establishment of native species, this section discusses the success of restoration at the other end of the restoration success continuum (Figure 6.1). At this level, successful restoration moves beyond the maintenance of species (and their role as individuals in biodiversity conservation) and moves toward the development and conservation of ecosystems and their role in biodiversity conservation. Conservation of individual species is still a dominant component at this level. However it is the role that individuals play within the system that is significant to the overall success of the system.

Restoration must restore both structure and function for successful biodiversity conservation at the ecosystem level. The literature surrounding the measurement of structure and function and how this may be achieved is rather confused (Berger 1991) and fewer studies have actually attempted such measurements. Any attempt to gain a detailed understanding of the structure and composition of a system, and the processes involved is constrained by a plethora of factors, from cost to the inability to monitor unmeasurable, abstract, ecosystem components.

The development of structure and function in the restoration study plots in this study are discussed separately. While both are vital attributes for a successful system and are interrelated, the existence of one does not mean we can assume the presence of the other. Both may develop at different rates and independently depending on the systems involved. The role of restoration in the processes of natural succession is discussed to tie together the development of structure and function within these systems to give insight into further development (if any). Finally the success of the plantings to restore structure and function, and the plantings role in succession are discussed together to give the holistic evaluation required to determine adequately if the restoration plantings in this study have been a success.

6.3.1 Ecosystem structure.

In this section I discuss whether ecosystem structure and composition have returned to the restoration plantings. Ecosystem composition is usually associated with ecosystem structure, being a closely interrelated component of a successfully restored system (Berger 1991; Cairns 1991; Westman 1991).

Ecosystem composition and structure can be defined in different ways. Composition is usually taken to refer to species presence and their relative abundances (often quantified with a diversity index). Structure is more complex, and includes aspects of vertical and horizontal patterning, as well as variability (heterogeneity) (Hobbs & Norton 1996).

Westman (1991) describes useful parameters of ecosystem structure such as; composition, absolute and relative abundances, gene frequencies, pattern of local and regional distribution, density, biomass, nutrient pools, topographic features, water quality and quantity, energy content, soil structure and soil/litter nutrient pools. Aronson *et al* (1993a) suggest that the following vital ecosystem attributes may be used to indicate

ecosystem structure; perennial species richness, annual species richness, total plant cover, above ground phytomass, beta diversity, life form spectrum, keystone species, microbial biomass and soil biota diversity.
While there appears to be some continuity between the above parameters, it is unlikely that all parameters are appropriate for all studies (Aronson *et al* 1993a). As Westman (1991) suggests, the number of parameters that could be monitored often exceeds available resources. However, many parameters are interrelated and those more difficult to measure may be estimated from the determination of others (Westman 1991; Aronson *et al* 1993a).

Due to the constraints associated with this study it is not possible to consider all of the above monitoring parameters. I agree with Hobbs & Norton (1996) who suggest that attempts to monitor restored systems are still in exploratory stages. While many studies suggest the evaluation of structure in restoration projects fewer describe components of structure and how to measure them (Majer 1989a; Andersen 1993; Berger 1993; Bradshaw 1983; Williams 1993; Chambers *et al* 1994; Hobbs & Norton 1996). Under the umbrella terminology of structure I will explore the following; structure, composition and heterogeneity (as defined by Hobbs & Norton 1996). While most studies of structure and composition usually evaluate the vegetation (McClanahan & Wolfe 1993; Hobbs & Norton 1996; Chambers *et al* 1994), ground invertebrates will be included in my analysis offering an additional 'dimension' to the measurable components of "structure and composition'.

The composition of the vascular vegetation at Kennedy's Bush differs in each study plot (Figure 4.5). Clearly vegetation composition hasn't been restored in the restoration study plots. However, as the restoration study plots age they do become compositionally more similar to the naturally regenerating and mature forest study plots. The composition of the study plots not only differs in terms of the species present, but also their abundances. For the restoration study plots these differences are partly due to the planted component of the plantings. *Olearia paniculata* contributes significantly to the vegetation composition of these study plots, however as other native species colonise these sites this initial planted component declines in significance. *Olearia paniculata* (the significant planted component of the restoration plantings) is not found in the naturally occurring forest study plots. Consequently, the composition of the restoration study plots is not similar to the naturally regenerating and mature forest study plots. The restoration study plots will only be compositionally similar to the naturally regenerating and mature forest study plots when the planted vegetation component becomes much less significant (ie physically removed or out grown by native species), provided the species recolonising the restoration study plots are similar to those found in the naturally regenerating and mature forest study plots.

The species richness, diversity and evenness of each vegetation layer (eg canopy, regeneration) differ between study plots (Figures 4.1 to 4.3). The naturally regenerating study plot shows the most similarity to the mature forest study plot in all of these measures. The great variability within each vegetation layer of the restoration study plots suggests that these study plots are in a state of development. Again, the planted component of the restoration study plots is likely to influence these diversity measures and possibly obscure any naturally occurring trends.

Although the floristic composition of the vegetation of the restoration study plots does not currently demonstrate plant composition similarities to the naturally regenerating and mature forest study plots, there are indications that this may change in the future. This will be further discussed in this section.

While the structural aspects of each study plot were not measured directly, structural similarities may be indicated using other measures. The amount of cover of the different vegetation layers indicates differences in the vertical arrangement of the vegetated components of each study plot (Figure 4.1b, 4.2b, 4.3b). For example, the regeneration layer indicates that the structural complexity at this height is greater in restoration 3. Similarly, this method implies that the mature forest study plot has the greatest structural complexity in the canopy layer. Comparing the structure of each study plot indicates differences between all the study plots, with restoration 1 being the least, and the naturally regenerating study plot the most structurally similar to mature forest study plot. As with the floristic composition of these study plots, it appears that the structure of the restoration study plots better resembles the naturally regenerating and mature forest study plots with increasing age. Other features of the study plots also illustrate these differences in structure. The height of the canopy of each study plot increases as the restoration study plots age, with the naturally forested study plots having the tallest canopies (Figure 3.1). The mature forest study plot has an additional structural component due to the presence of emergent trees, whereas the naturally regenerating study plot has a subcanopy vegetation layer not found in the other study plots. In addition, the mature forest and naturally regenerating study plots have a greater forest floor deadwood component in comparison with the restoration study plots. This indicates that the deadwood component of these study plots may also be greater in other layers of the system (eg. canopy) as the relative abundance of deadwood on the forest floor (presumably having fallen from above at some point) may indicate the abundance of dying or dead branches, for example, higher in the study plot.

Compositional heterogeneity may be measured using a variety of techniques. Lapin & Barnes (1995) suggest the use of Jaccards index to measure compositional heterogeneity. Compositional heterogeneity based on the floristic similarity of study plots in this study (Table 4.2) shows that the two oldest restoration study plots are as compositionally heterogeneous as the naturally regenerating and mature forest study plots. Compositional heterogeneity is the only aspect of structure where the vegetational components of any of the restoration study plots are similar to the naturally regenerating and mature forest study plots. This means that the variability in species composition within a study plot is similar in these four study plots, despite those differences in species and abundances between study plots.

Andersen (1993) and Simmonds *et al* (1994) measured the composition of ant and spider communities respectively, to assess the success of rehabilitated mine restoration projects. Both studies reported trends in the development of invertebrate community composition associated with habitat development (a function of site age) (Andersen 1993; Simmonds *et al* 1994). Similar trends are shown with respect to the invertebrate data in this study (Figures 5.5 to 5.7, Table 5.2c, 5.4c). The composition of the ground invertebrate communities differs with respect to all study plots and is correlated with the vegetation and study plot age. Analysis of beetle and spider groups independently mirror this result, suggesting no invertebrate component of the restoration study plots is compositionally similar to the naturally regenerating and mature forest study plots. The combined invertebrate data (Figures 5.5b to 5.7b) show similar trends to the vegetation composition.

Comparisons of vegetation and invertebrate structure and composition suggest that none of the restoration study plots has successfully restored these ecosystem components. While a significant component of the observed differences in the composition of the vegetation may be attributable to the strong presence of planted species in the restoration study plots that are not found in the naturally occurring systems, the study plots are structurally different. The two oldest restoration study plots have developed similar compositional heterogeneity to the naturally occurring study plots. Therefore the restoration study plots are capable of restoring aspects of structure. This and the trend of increasing structural and composition development with restoration age suggests that the other components of structure and composition may also be restored, given time.

6.3.2 Ecosystem function.

Ecosystem function as well as structure must be returned to a site if restoration is to be considered successful. While many authors (eg. Majer 1989a; Cairns 1991; Aronson *et al* 1993a; Main & Lambeck 1993; Saunders *et al* 1993; Chambers *et al* 1994; Hobbs & Norton 1996) consider the importance of function in a restored ecosystem, fewer describe factors that contribute to a functioning system and how to monitor them.

Hobbs & Norton (1996) suggest ecosystem function is the performance of basic ecological processes such as energy, water, nutrient transfer. Aronson *et al* (1993a) list 11 vital ecosystem attributes related to ecosystem function. These include biomass productivity, soil organic matter, maximum available soil water reserves, coefficient of rainfall efficiency, rain use efficiency, length of water availability period, nitrogen use efficiency, microsymbiont effectiveness and cycling indices (Aronson *et al* 1993a). Such attributes may be complicated and difficult to measure (Westman 1991). Aronson *et al*

(1993a) do not suggest which, if any of these attributes are most important. However, the relative importance of attributes is likely to depend on the system, the level of degradation and the environment.

Westman (1991) suggests the following parameters as indicators of ecosystem function; productivity/growth rates, nutrient flux, pollutant flux, natality/mortality rates, migration, fire frequency/intensity, hydrological flow, soil movement, radiation flux. Ideally more than one parameter should be chosen (from a list of functional (above) and structural parameters) for monitoring each of the major biotic and physical components of a site to achieve a representative indication of restoration performance (Westman 1991). While Westman (1991) fails to indicate which parameters may be important, he does imply that it is not necessary to monitor all parameters to determine restoration performance. It appears that parameters should be chosen with low levels of intercorrelation to maximise information content and should be based on what is considered appropriate for each site (Westman 1991).

Chambers *et al* (1994) suggest that comparisons of structure between restoration and reference site can be valuable indicators of function and are relatively easier to assess. For example, standing crop production (gm^{-2}) can be measured in place of productivity ($gm^{-2}\ yr^{-1}$), density may be measured instead of population turnover and functional guilds can be used to indicate changes in function caused by the presence or absence of species (Chambers *et al* 1994). However, Westman (1991) suggests that it is not valid to assume that the restoration of structure will achieve the restoration of function, as structural and functional attributes develop at different rates. Armstrong (1993) suggests that the identification of functional groups is constrained by limited information on relationships between species and function and the simplification of complex interactions may be made by classifying functionally equivalent species into functional groups.

These examples illustrate the lack of continuity between ideas for measuring ecosystem function. While the above parameters are not necessarily invalid, monitoring all parameters would be inappropriate due to cost and time constraints. It appears that some of the parameters suggested may be more appropriate for monitoring some projects than others. In this study I take the approach of monitoring those parameters that give the best indication of ecosystem function in consideration of the cost, time and knowledge constraints that I am subject to.

Ecosystem function can be restored in the absence of ecosystem structure (Berger 1993; Cairns 1993a; Brown 1994; Hobbs & Norton 1996). Any system will have function to some extent and will continue to do so even under extreme levels of stress. The presence of species ensures this as each individual plays a role in the system in which it lives. A system may cease to function only when it is void of any species.

It is important to recognise for restoration projects that ecosystem processes occur at a minimum scale and each process does not necessarily occur at the same scale (Westman 1991). Therefore the size of a site will influence the functional processes of a system (Westman 1991; Saunders *et al* 1993). Due to effects associated with fragmentation, ecosystem function is also influenced by the surrounding environment or landscape (Cairns 1991; Westman 1991; Fry & Main 1993; Haila *et al* 1993; Saunders *et al* 1993; Hobbs 1994; Naveh 1994; Norton *et al* 1995; Hobbs & Norton 1996). Westman (1991) suggests that these should be considered when choosing restoration criteria. Post restoration management plans should also recognise that certain key ecosystem processes will be compromised by fragmentation.

While it is implied that ideally we should be monitoring all parameters associated with ecosystem function, this was not the approach taken in his study due to the constraints mentioned earlier. The approach used in this study was to look for indications that similar functional processes were occurring in the restoration study plots and the naturally occurring study plots. The presence of certain processes may indicate that other processes are occurring in the system as all processes within a functioning system are interrelated (Westman 1991; Aronson *et al* 1993a). Westman (1991) identifies the need for research to identify easily measurable ecological parameters that may serve as indicators of a larger range of processes, when recognising that the number of possible parameters to monitor may greatly exceed available resources.

The presence of regeneration in all three restoration study plots indicates that the functional processes that initiate regeneration are present. In the youngest restoration planting, regeneration is likely to be only from the dispersal of propagules from outside the area. The two older restoration study plots however display regeneration by propagules from both within and outside the study plots. Regeneration of seedlings from plants within the restoration study plots indicates that other important ecosystem processes are occurring. For example, for plant species to fruit the processes of pollination must be present. The presence of regenerating plants within the restoration study plots indicates that the ground or litter layer has developed and is functioning adequately. Litter accumulation is also an important process and many elements of a forest system are dependent on the functional aspects of this layer (eg. mycorrhizal associations, chemical and nutrient levels, moisture regimes).

Invertebrates are important indicators of ecosystem function (Majer 1989a,b; Andersen 1990,1993; Williams 1993; Simmonds *et al* 1994). While ants (Andersen 1993) and spiders (Simmonds *et al* 1994) have been suggested as good indicators of functional activity, Hutcheson (1990) suggests the use of Coleoptera. Coleoptera utilise most trophic niches, comprise greater than 40% of all insect species, are generally representative of insect fauna 'richness' (Hutcheson 1990). Forest floor dwelling representatives of the family Coleoptera were used in this study as indicators of functional processes.

The presence of detritivore, predator and live plant feeder guilds in the three restoration study plots (Figure 5.4) suggests that the processes of decomposition, predation and herbivory are present. While the number of beetle species is similar for the restoration plantings, the number of individuals differs (Figure 5.4). While slight differences in abundances and those species present (Figures 5.4, 5.5) implies that that the extent to which the restoration study plots function is not identical, the presence of these guilds implies similar functional processes are occurring successfully. As suggested, the presence of one process may indicate the existence of others. Williams (1993) suggests that decomposition will contribute to successful nutrient cycling and plant establishment.

The absence of live plant feeders from the mature forest study plot (Figure 5.4) is probably a function of sampling height rather than a reflection of community composition. I suggest that as the dominant portion of edible vegetation (ie sub canopy and canopy) moves to a greater height with the forest canopy, the herbivore component will follow and will therefore be out of the range of pitfall trapping.

Although complex and intricate monitoring of ecosystem processes, as suggested by Aronson *et al* (1993a), has not been possible in this study, the presence of regenerating plants and invertebrate guilds infers that the two oldest restored study plots have functional aspects similar to the naturally regenerating and mature forest study plots. Assuming that the naturally regenerating and mature forest systems are successfully functioning, based on the presence of key ecosystem processes, I suggest that the presence of these key processes in the restoration study plots implies that they too are functioning at a similar level to the natural forest systems. While slight differences in abundances and species may be interpreted to suggest that those processes identified may be occurring at different levels (eg greater pollination at one site compared with another), this does not mean that the ecosystem processes occurring at the oldest two restoration study plots are different to those in the naturally regenerating and mature forest study plots. One recognisable difference in the three restoration study plots and the naturally regenerating and mature forest study plots is the time available for the development of the litter layer. The naturally regenerating and mature forest study plots have better established and developed litter layers than the restoration study plots. Despite the importance of this layer to forest systems, differences do not appear to result in the exclusion of some processes from any sites. It may be that such differences only become apparent in times of intense ecosystem stress (eg. large scale disturbance).

To observe the extent to which the functioning systems in this study may differ would require much greater time and expertise than was available. For example, improved access and time with greater taxonomic resources may have enhanced the understanding of the roles that individual invertebrate species play in the functioning of these systems (provided the appropriate knowledge on the those species were available). It is clear from this study and others, that while the functional state of restored ecosystems is vital for the success of the system, methods and concepts for evaluating those processes involved are not straight forward.

6.3.3 <u>The role of restoration in natural succession.</u>

The current paradigm in restoration ecology involves returning a degraded system to a desired state by accelerating or reinstating successional processes (Ashby 1987; Uhl 1988; Bradshaw 1989, 1987; Majer 1989a; Luken 1990; Hobbs & Norton 1996). It is suggested that in all but the most degraded sites natural successional processes should return a system near to its natural state with minimal human interference (Norton 1991; Aronson *et al* 1993a,b). In many cases restoration is intended to return a site to near its pre disturbance condition faster than it would without human interference (Norton 1991; Andersen 1993).

In the New Zealand situation, a number of pathways are possible for secondary succession to native forest (Williams 1983; Wilson 1994). Introduced species, gorse, broom and elder have dominated early successional stages as a consequence of European arrival (Williams 1983). Williams (1983) suggests a broom-elder-native forest (mahoe) may take around 50 years from the establishment of broom and is comparable with kanuka to mahoe succession in Wellington studied by Druce (1957). Historically kanuka-mahoe succession was the dominant pathway to podocarp forest with kanuka invading after fire and associated disturbances (Allen *et al* 1992). While kanuka can invade bare ground and lightly grazed, short-stature pasture (Allen *et al* 1992), it and other woody species have little ability to establish in ungrazed pasture (Allen *et al* 1992; Wilson 1994). Once kanuka has established the canopy may remain closed for up to 70 years, with any other seedlings under this canopy either dying or being suppressed (Allen *et al* 1992). The opening up of the canopy allows the establishment of other woody species, with podocarps establishing at later stages (Esler & Astridge 1974). Molloy (1975) has indicated that these successions through kanuka to forest may take up to 200 years or more.

The high correlation of vegetation with study plot age (Figure 4.5, Table 4.4b) illustrates that the vegetation of the restoration study plots is undergoing successional processes in the direction of the naturally regenerating and mature forest study plots. The high correlation of invertebrates (and spiders) with study plot age and vegetation (Table 5.2c & 5.4c) mirror this successional trend. However, the invertebrates are likely to be responding to the vegetation of the study plots, with the vegetation of the study plots a function of study plot age. The level of both the invertebrate and vegetation correlation with study plot age and the relative absence of any other measured environmental factor indicates that study plot age is the determining factor in structuring invertebrate and vegetation compositions.

If it is assumed that the regenerating tree species will dominate the future canopy, observations of the regenerating vegetation may be used to indicate the future canopy composition. The regenerating vegetation of the restoration and naturally regenerating and mature forest study plots is very similar (Figure 4.7a,b). Therefore, an ordination of each study plot's regenerating vegetation and canopy vegetation (Figure 4.8) should indicate how closely the future canopy of a study plot will resemble the current canopy. Information from the ordinations of study plots based on this assumption (Figure 4.8) suggests that the future canopy of the mature forest study plot will more closely resemble the current canopy than is the case with any of the other study plots. Such an observation provides the basis for two suggestions regarding the future vegetation of the study plots in this study. Firstly, it appears that the canopy vegetation of the mature forest study plot will replace itself. Although an absence of kahikatea seedlings (pers. obs.; Burrows 1994b) suggests that kahikatea will be absent from this canopy. Secondly, if the regeneration of tree species is similar in all study plots then the future canopies of the three restoration study plots and naturally regenerating study plot will resemble the future (and present) canopy of the mature forest study plot (Figure 6.2). In addition these two assumptions imply that the vegetation of the mature forest study plot is a self perpetuating community and that it is appropriate to assume that restoration to a pre disturbance condition in the Port Hills area could use Ahuriri Scenic Reserve as a potential model for restoration success.

Hobbs & Norton (1996) suggest that systems may potentially follow alternative successional pathways depending on the combination of management, climatic and biotic factors. Thus the outcome of any aspect of restoration may differ, being potentially influenced by differing locations and times that it may have been initiated. Since Diamond (1975) introduced the concept of assembly rules there has been much theoretical discussion regarding the assemblage of communities (Gibson *et al* 1985; Schuster & Hutnik 1987; Cairns 1989; Burrows 1990; Wilson *et al* 1995; Drake 1990; Drake *et al* 1990; Luh & Pim 1993). Yet we are just beginning to understand those processes by which communities assemble (Hobbs & Norton 1996). While such assembly rules have been suggested to be valuable in predicting the outcome of restoration projects (Luh & Pim 1993), Hobbs & Norton (1996) suggest we are far from making predictions on the outcome of adding species in particular combinations and orders.

The evidence presented here on the future of the restoration and naturally forested study plots in this study may be regarded as circumstantial. Despite suggestions that community development may be unpredictable, the evidence presented here suggests that the restoration study plots will develop toward a community similar to that of the mature forest study plot. It is important to recognise however, that the resulting communities will be similar, but not identical, to the mature forest study plot.

It seems only practical to indulge in restoration projects if intervention will enable the resulting community to develop considerably faster than what would occur by natural processes. While direct comparisons between the development of natural and restored communities seemed to be the best way to test this, the relative development of respective communities may provide an adequate indication.

Williams (1983) and Allen *et al* (1992) suggest time periods of 50-70 years for the colonisation of native forest plant species beneath early successional canopies of (gorse-broom-elder) and kanuka successional pathways. Therefore the relative proportion of regeneration (Figure 4.7) and overall compositional vegetation development (Figure 4.5, 3.1) of the two oldest restoration study plots (30 and 35 years old) suggests that the restoration study plots in this study are providing conditions suitable for recolonisation and establishment faster than those occurring under natural processes.

It appears that the high component of *Olearia paniculata* is obscuring the extent of the restoration study plot's success. As mentioned earlier such a dominance of this species in the canopy is distracting in terms of composition from the relative success of recolonisation and establishment.

Restoration may provide a huge head start on natural succession by immediately shifting a system from one metastable state to another. While there is little evidence suggesting the period a grassland community on the Port Hills may remain uninvaded by bracken and other species, Allen *et al* (1992) suggest that these areas may remain in grassland states for several decades. The absence of forest species (or any other species other than grassland) in the grassland study plots and suggestions by Allen *et al* (1992) and Wilson (1994) regarding the difficulty of tree species to establish in tall grasslands indicates that these study plots could remain in grassland states for large periods of time in the absence of disturbance.

6.3.4 Have the restoration plantings at Kennedy's bush been successful?

In terms of the goal described in chapter two, the restoration of self-sustaining functional native forest ecologically suitable for the area, the three restoration study plots at Kennedy's Bush appear to have been unsuccessful. The restoration plantings have failed to restore structure and composition. However the two older sites in particular, have restored those functional processes identifiable from natural systems. Although such processes have been identified, it has not been possible to identify accurately the extent to which these processes are occurring in these systems and hence the level each system is functioning at.

The restoration plantings appear to be self-sustaining in that they appear not to require further human input. Yet they appear to be developing toward the natural forest systems of the area. While the restoration plantings are far from developing to a self perpetuating state, indications above suggest that under the current conditions this will eventually occur.

Although the restoration plantings have failed in terms of the goal set earlier (chapter two), they are successful in terms of native biodiversity conservation. Forest cover has been restored, and this has provided habitat for native forest species, both vegetation and animal. In addition to creating new habitat, the restorations have provided areas that may have greater asthetic appeal by providing relief from grassland and increasing continuity with the naturally occurring forest vegetation.

While the restoration plantings can be viewed as unsuccessful, the future looks promising. Results from this study indicate that given time the study plots should develop toward the natural forest systems of the area and may attain those ecosystem features required to satisfy the restoration goals.

This study draws attention to the future of restoration on the Port Hills. While the environment has been heavily modified, it does not appear to be as difficult to reestablish native forest vegetation as suggested by Saunders *et al* (1993). For example, the results from this study suggest that, due to relatively close seed sources and existing remnant patches of native forest combined with adequate dispersers, moving from the grassland state may be the most difficult aspect to restoration in this area. Once this initial step has been initiated the natural processes of the area appear to be strong enough to move a planting in the direction of native forest, as suggested by this study.

One of the most important features illustrated by this study is the inherent difficulty surrounding the measurement and evaluation of biological systems. While there are sufficient problems to be encountered when trying to assess the state of a natural system, trying to predict the outcome of a restored system considering the confused state of knowledge surrounding such processes is a very difficult task.

The restoration plantings in this study are relatively old in comparison with many other studies evaluating restoration success. While this provides an exciting opportunity to examine older systems they were initiated without the knowledge and insight that would be available if the same areas were to be restored today. This will always be the way with science. Yet it is likely that we will wonder whether restoration plantings would be successful any faster if initiated with current knowledge.

While restoration is currently under pressure to be a 'proper' science, I believe that the intense evaluation (especially as suggested by Aronson *et al* 1993a,b) of restoration plantings may potentially distract from what we should be trying to achieve in the first place. Restoration plantings may always be deemed unsuccessful if they may never attain the stringent scientific goals set by us, or fail to return exactly to the predisturbance condition as is often expected. I suggest that the failure of these plantings to successfully restore structure and composition may in part be due to my definitions of these attributes being to restrictive.

More importantly, restoration can force us to acknowledge our past mistakes that lead to environmental degradation. While current restorations may fail to return those systems we have so heavily degraded, the process of restoration can remind us of the great beauty and value in our natural systems. Hopefully our restoration efforts will inspire us to show greater consideration and appreciation for those remaining intact natural systems. Although some may argue that restored areas may never be as good or natural as the natural systems before them, those involved in restoration with their feet firmly planted on the ground would never expect such miracles. What's gone is gone, yet the alternative too restoration, bare earth, is the least attractive option.

CONCLUSIONS

(1) The three restoration plantings at Kennedy's Bush have successfully facilitated the recolonisation and establishment of native forest species (plant and invertebrate), and thus are successfully conserving native biodiversity.

(2) It appears that those species used in the initial restoration planting at Kennedy's Bush may not greatly influence the outcome of the restoration.

(3) Introduced birds and possibly other non-native species may play an important role in the success of these restoration systems.

(4) The three restoration plantings have failed to restore structure and composition.

(6) The two oldest plantings have restored those ecosystem processes found in the naturally occurring forest plots, suggesting they have restored ecosystem function.

(6) The restoration plantings appear to be able to facilitate the above components of restoration success faster than would occur naturally in the absence of human intervention.

(7) This study suggests that the restoration plantings in the future will restore ecosystems similar to that found at Ahuriri Scenic Reserve.

(8) This study illustrates the inherent difficulties surrounding the monitoring of ecosystems, particularly those attributes associated with ecosystem function.

(9) This results from this study suggest that restoration on the Port Hills has a bright future, and restoration can be undertaken with relative ease and high chances of success. It appears that establishment may be the only barrier to further ecosystem development.

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147pp

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